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Ecology of Savanna Ecosystems in Indonesia

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ABSTRACT

Tropical savannas in South East Asia are important yet understudied ecosystems. In fact, the description of savanna is limited in the Indonesian Archipelago, a region which, based on climate alone, would be expected to have mostly forest. In this thesis, I compared and contrasted the vegetation characteristics of four savannas in the wetter part of the Indonesian archipelago (Java – Bali – Lombok) to understand how fire and grazing influence their dynamics, and searched for evidence of savanna origins, maintenance, invasion by exotic/woody species and possible ecosystem transitions. Vegetation surveys, remote sensing techniques and statistical models were used to spatially and temporally analyse the savanna community composition and the environmental variables and disturbances that influence their structure.

Results showed that there are distinct elevation gradients (along with related climatic factors such as temperature and precipitation), as well as fire regime gradients, linked to tropical savanna community composition across Java, Bali and Lombok Islands. These compositions are characterized by different sets of species, and where invasive alien species are becoming significant components of the ecosystems. Lack of prescribed fire and a range of invasive species threaten to convert savanna at Bali Barat and Alas Purwo into secondary forests or shrubland, whereas the presence of forest pioneer/edge species within the savanna at Rinjani suggests successional change from grassland to forest may occur in the absence of future fires (although the role of soil, topography and microclimate in maintaining grass dominance needs also to be further explored). Compared to the others studied, the savanna in Baluran National Park has characteristics of being relatively old and persistent rather than one being created and maintained via recent human conversion of forests.

Overall in Indonesia, there is much less savanna compared to forests, hence it is expected that a greater percentage of savanna is burned. Using remote sensing analysis, I confirmed that approximately 2% of savanna/open vegetation had burned over a 14 year period, whereas only 0.8% of forest has burned across Indonesia. The extent and frequency of burning is mostly associated with annual Southern Oscillation Index (SOI). Most burning occurred in years when the SOI sustained negative SOI values, which generally means drier conditions across South East Asia.

I also developed species distribution models for the main invasive alien species of the savanna ecosystems studied, *Acacia nilotica,* to establish its invasion potential, both locally in Baluran National Park and regionally in other parts of Indonesia. *Acacia nilotica* was different from the other invasive species studied, in that it is promoted by herbivory, and possibly also by fire. It appears that spatially, *A. nilotica* is rapidly advancing into the savanna of Baluran National Parks where it was observed that over fourteen years the savanna size has decreased (-1,361 ha), whilst the *A. nilotica* stand has increased in area (+ 1,886 ha). It was demonstrated that fire and grazing play an important role in this invasion. Results also show that global climate change is likely to increase the potential distribution of *A. nilotica* in Indonesia and the area at risk of invasion. By year 2045, *A. nilotica* has potential to spread across much of the eastern parts of Indonesia. As fire and grazing are common to savannas of eastern Indonesia, they are likely to facilitate its invasion into these areas.

In summary, I have shown that savanna plant community in Indonesia is formed and maintained by interactions between climatic factors, fire regime and grazing. Invasive species were also present in the studied savannas such as *Chromolaena odorata*, and *Lantana camara*. These invasive species together with forest pioneer/edge specialist species (*Ficus septica, Laportea stimulans, Melastoma polyanthum, Nauclea orientalis, Rubus rosifolius*), may also be increasing in absence of fire and also may be leading to change of state from savanna to dense woody vegetation. Absence of fire seems to be changing structure and floristic of savanna vegetation which has implications for savanna species including rare fauna such as Jalak Bali/Bali Starling (*Leucopsar rotschildi*) and Javan Banteng/Wild Java cattle (*Bos javanicus* subsp. *javanicus*). Results from this thesis showed that Bali Starling range in Bali Island has shrunk to remaining small patches of fire-induced open shrub and savanna woodland found below an elevation of 150–175 m in the north-east part of peninsular Prapat Agung of Bali Barat National Park.

The description of the savanna dynamics presented here provides further evidence of the complexity of the savanna ecosystem and its susceptibility to change as a result of changing fire regimes and invasion by invasive species. A greater understanding of the possible ecosystem processes driving the dynamics of the savannas will assist in the formulation of successful savanna management strategies at local and regional scales.

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Signed: Sutomo

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This thesis is dedicated to my mother who passed away during my candidature, may Allah grant you *Jannah*, and we love you and miss you so much!

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Perth, October 2017

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STATEMENT OF CONTRIBUTIONS OF OTHERS

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The materials presented in this thesis are a synthesis of my own ideas and the works undertaken by myself in consultation with my supervisors, Dr. Eddie van Etten and Dr. David Goodall. I led the development of the questions and hypotheses at all stages, undertook the fieldwork and analyses, and completed the writing. The above list of co-authors contributed in one or more of the following ways to the thesis components published in journals and proceedings: data collections, provisions of unpublished datasets, analyses, or development and editing of manuscripts.

FORMAT OF THESIS AND PUBLICATIONS ARISING FROM THIS RESEARCH

This thesis is presented in the "thesis with publications" format. It is composed of a General Introduction, six Research Chapters and a General Discussion to integrate the research chapters and make final conclusions and recommendations. The research chapters in this thesis are written in the style and format of stand-alone papers to facilitate submission to peer-reviewed journals; hence there is some repetition of several parts mainly in the introduction, sites descriptions and methodology throughout. Figure and table number throughout the thesis is restart for each chapter.

Citations details of publications that have arisen from this thesis are as follow:

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- Part of chapter 4:
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- Part of chapter 5:
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  (http://biodiversitas.mipa.uns.ac.id/D/D1701/D170114.pdf)

Sutomo & van Etten E. (2017) Species distribution model of invasive alien species Acacia nilotica for Central-Eastern Indonesia using Biodiversity Climate Change Virtual Laboratory (BCCVL). *Tropical Drylands* **1**, 36-42. (Journal). (https://smujo.id/td/article/download/1644/1580/)

- Parts of chapter 6:
  Sutomo & van Etten E. (2016) Effect of fire and digestion by herbivores on seeds of the exotic invasive species *Acacia nilotica* from savanna at Baluran national parks Indonesia. *Australasian Plant Conservation* **25**, 20-1. (Bulletin)

- Sutomo, van Etten E. & Priyadi A. (2015) Do Water Buffalo Facilitate Dispersal of Invasive Alien Tree Species *Acacia nilotica* in Bekol Savanna Baluran National Park? In: *Second International Conference on Tropical Biology* (eds E. K. Damayanti and J. C. Fernandez) p. 155. SEAMEO BIOTROP, Bogor. (Conference proceedings)
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<u>In Review:</u>

- Chapter 3:
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- Part of chapter 5:
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<u>Non peer-reviewed publications:</u>

- Sutomo. (2014) *Invasion of Exotic Species Acacia nilotica in Savanna Ecosystem of Baluran National Park East Java Indonesia*. Interlude, Yogyakarta. (Book)

- Sutomo. (2017) Evicted from its own home: Bali Starling (*Leucopsar rotschildii*). AIPSSA, Perth. (AIPSSA 3 Minutes Thesis Competition presentation video, Curtin University 29 April 2017).

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TABLE OF CONTENT

LIST OF FIGURES

CHAPTER 3

CHAPTER 4

CHAPTER 5

CHAPTER 6

CHAPTER 7

CHAPTER 8

LIST OF TABLES

CHAPTER 7

Chapter 1: General Introduction and Study Site Description

1.1. SAVANNA DISTRIBUTION IN INDONESIA

Savannas are ecosystems mostly confined to tropical and subtropical regions that are characterized by a continuous cover of C_4 grasses that have different characteristics based on seasonality and where woody plants are also an important feature, but with generally sparse cover and never a closed canopy (Frost *et al.* 1986). Tropical savannas cover over 20% of the Earth's land surface, with the largest coverage in Africa, Australia and South America. They include widespread and well-studied savanna ecosystems, such as *Eucalyptus* woodlands in northern Australia (Burrows *et al.* 1991; Werner 1991), and the miombo woodland of southern Africa (Isango 2007). Only approximately 10% of savannas occur in India and South-East Asia (Bond and Wilgen 1996; Furley 2004; Werner 1991). One of the earliest descriptions regarding tropical savanna ecosystems in South East Asia, especially India, Burma and Thailand, was done by Kurz (1876). One of Kurz's study sites was in Thailand where open savanna sites are usually topographically drier, such as plateaux in the rain shadow of mountain ranges. Common species in these savannas were *Imperata cylindrica*, *Panicum repens*, *Saccharum spontaneum*, *Sorghum halepense*, *Vitiveria zizanoides*, *Chromolaena odorata, Acacia cathecu, Acacia siamensis, Careya arborea*, and *Pterocarpus macrocarpus*. Where such savanna occur in SE Asia, they often provide important habitat for large browsing and grazing mammals such as Barking deer (*Muntiacus muntjak*), Sambar deer (*Cervus unicolor*) and Banteng (*Bos javanicus*) (Kurz 1876).

Savannas in Indonesia are located from west to the east across the archipelago. These savannas occur across a steep gradient of rainfall (Monk *et al.* 2000; Whitten *et al.* 1996). Savanna in West Java receive an annual average rainfall of 2,940 mm, with 68.9% of this rainfall occurring during the wet season which lasts from October to April (Rosleine and Suzuki 2013). On the east side of Java Island, rainfall is lower. Savannas here have annual rainfall average of 1,500 mm, with the wet season being more pronounced (80% of total rain), usually occurring from November to April, with the dry season occurring from May up to October (Hakim *et al.* 2005). In Bali and Lombok Islands, savannas receive an average rainfall of 1,200 – 1,500 mm. In East Nusa Tenggara, the average annual rainfall drops to 800-900 mm (Gunawan 2010; Monk *et al.* 2000).

Indonesian savanna research is currently lacking with only a few studies done (Monk *et al.* 2000; Whitten *et al.* 1996). Two of the foremost early studies on Indonesian flora and vegetation was that of Backer & van den Brink (1963) and van Steenis (1972). However, these books focussed on the mountain flora of Java Island and its grasslands, and savanna was not discussed. Fire, according to van Steenis (1972) was presumed as the major factor that drove the existence of grasslands on Javan mountains, especially in East Java which is subject to a more pronounced dry season (less rain in the dry season and longer period of dry season) and lower overall annual precipitation compared to other parts of the Island. Sumardja and Kartawinata (1977) described savanna in Pangandaran Peninsula, West Java. They reported that Batu Meja and Karang Pandan savanna sites in Pangandaran had been abandoned around 1957, following which it became young secondary forests of *Decaspermum fruticosum*. Whitten *et al.* (1996) mention the existence of savanna in Baluran, East Java and Bali Barat in Bali. However they did not discuss this thoroughly. Savannas at these sites have *Acacia* as their main woody species, and in Bali Barat there is added dominance by Lontar trees (*Borassus* spp.) (Whitten *et al.* 1996). In the beginning of the 21st century, savanna research in Indonesia started to gain traction. Sabarno (2002) described species composition of the Baluran savanna where the thorny *Acacia nilotica* is recognized as alien invasive weed that has been threatening the existence of the Baluran Savanna. Rosleine and Suzuki (2013) studied secondary succession on abandoned savannas in Pangandaran peninsula where *Imperata cylindrica* and *Chromolaena odorata* are obnoxious invasive species. They also state that only about 12 ha of savanna remain in Pangandaran, whilst 24 ha have shifted to secondary forest. Studies are lacking for other savanna in Java such as the Alas Purwo. Similarly, savanna on Bali Island also remains understudied.

Savannas in East Nusa Tenggara Islands (NTT) seem to have received better attention (Fisher *et al.* 2006; Russell-Smith and Edwards 2006; Russell-Smith *et al.* 2012). In many seasonally-dry places of NTT, savannas and grasslands are the dominant vegetation type. There are four types of savannas commonly found in NTT, namely: *palm* savanna, *eucalypt* savanna, *Acacia* savanna and *Casuarina* savanna. These savannas are named after the main tree species which characterize each savanna. Although there are some descriptions of the savannas in NTT, they are however generally inadequate when compared to the area of NTT covered by savannas (Monk *et al.* 2000). Species composition of savanna in NTT was studied by Auffenberg (1981) and described in terms of dominant woody species. *Borassus flabelifer* (Palmae) dominated the tree layer of savanna at Komodo Island, Rinca Island and the northern and southern coast of Flores Island up to an altitude of about 400 m above sea level (asl), whereas *Ziziphus mauritiana* was found growing from sea level up until 500 m asl. In terms of the role of fire in NTT savannas, literature is lacking, although Fisher *et al*., (2006) did map fire scars over a two year period at several savanna sites in Sumba and Flores Islands.

The composition of savanna plant communities in Indonesia appear to be rather different from place to place and is worthy of further investigation of what factors may be affecting these differences. For savannas located elsewhere in the world, the answer is clearer. For example, in Africa, the structure and composition of the savanna ecosystem depends largely on rainfall, level of quantification of large vertebrate herbivory and the frequency of fires that burn the grass swards (Bond and Wilgen 1996; van Wilgen *et al.* 2000). Whereas, in Australia, there is strong relationship between climate of savanna sites and their fire frequency which, in turn, affects the tree basal area and thus the continued existence of the savannas (Lehmann *et al.* 2014b). The difference between Australia and Africa savanna structure and composition could also due to differences in grazing as it is noticeable that there are fewer grazers in Australia compared to Africa, however Lehman *et al.* (2014b) did not include this in their hypotheses. In general, savannas are controlled by the interplay of various factors and occur across large gradients of environmental conditions (Belsky 1994; Murphy 2008; Zimmermann 2008).

1.2. GENERAL TYPES OF SAVANNAS

There are currently three general types of savannas recognised. A climatic savanna is the term used for savannas that are the result of climate conditions. Edaphic savannas are primarily the result of soil conditions, and derived savannas are the result of forest clearing/disturbance by humans (Ford 2010; Murphy 2008; Scheiter 2008). Climate seasonality as well as disturbances are the main factors that confine savanna ecosystems (Sankaran *et al.* 2004; Scheiter 2008). Seasonality manifests itself mainly in terms of major differences in rainfall between wet and dry seasons. Dry seasons mean species need to cope with up to several months of drought which contributes to more open vegetation, as well as dead grass, which contributes to fuel load and therefore fire. Such conditions are less likely to occur in wet tropical regions with less pronounced seasonality. Savannas located at high rainfall sites are typically dominated by woody vegetation and in the absence fire can have almost closed tree cover that would outcompete and suppress grasses species (Rosleine and Suzuki 2013; Scholes and Walker 1993). Thus, in these sites, regular fire is important to establish grass-tree coexistence (Sankaran *et al.* 2003). In contrast, savannas at drier sites, woody vegetation is mostly in low abundance whereas grasses are the dominant component. Arid savannas have resources limitation (water competition) and lower biomass/fuels, and therefore fire is perhaps less influential for grass-tree coexistence in this type of savanna (Sankaran *et al.* 2005). Edaphic factors can also have a profound effect on savanna. In a study of savanna in Australia, for example, Williams *et al.*, (1996) found that tree cover and basal area decreased with increasing clay content, and other studies in other parts of the world have also found that woodland cover decreased with increasing clay content (Ben-Shahar 1991). In addition, vegetation type was found to be associated with soil nutrients levels in a South African savanna ecosystem, where high grass cover was associated with lower nutrients, whereas the reverse was true for trees species (Ben-Shahar 1991). Since for thousands of years both savannas and grasslands around the world have been manipulated and disturbed by humans for their benefit, it can be difficult in some areas at times to separate the influence of humans from that of natural processes in the formation or maintenance of these ecosystems (Ford 2010). Most recent literature would suggest that most of world's savanna have been maintained by fire whether it anthropogenic or natural (de L. Dantas *et al.* 2013; Ratnam *et al.* 2016).

1.3. DYNAMICS OF SAVANNA VEGETATION

In savanna vegetation dynamics, fire is a non-selective ecological driver where the degree of impact is mainly determined by the size and intensity of the fire, and also the susceptibility of individual species (Shannon *et al.* 2011). Woody species biomass is reduced when frequent, high intensity fires occur (Bond and Wilgen 1996; Shannon *et al.* 2011). Another important factor that drives savanna vegetation dynamics is grazing (Staver *et al.* 2009). However, although both factors are recognized as important drivers, not much research has been conducted to address the combined effect of these factors (Staver *et al.* 2009; van Etten 2010). One of the few to study such combined effects is Staver *et al.* (2009) who studied the effect of fire and grazing interaction to suppress tree density in African savanna. They found that combination of grazing and fire had much greater effects on tree density than either acting alone. Only fire and grazing together were able to prevent increases in tree density. This interplay between fire and grazing can lead to the phenomena of retreat and advance of savanna ecosystems over time, especially in terms of movements of the often sharp boundaries between savanna and forest areas (van Etten 2010). Forest-savanna boundaries are of important research value as they provide windows of opportunities to study the spatial patterns and dynamics of vegetation transitions. Since boundaries are rarely in a stable state (due to natural or anthropogenic disturbances), it is interesting to monitor them as they could point to advance or retreat of one ecosystem against another that could underline changes in the environment, such as changes in fire regime (Furley 2004; Furley *et al.* 1992; Hennenberg *et al.* 2005; Porensky 2011). Savanna is often considered as an alternative state within a state and transition framework (Gillson and Ekblom 2009; Twidwell *et al.* 2013). Van Langevelde *et al.* (2016), describes the feedback loop between fire and grazing effect on grasses (fuel load) and woody biomass. Intense grazing would cause a drop off in grass biomass which then leads to reduced fuel load that makes fire less severe and, thus, less damaging to trees, therefore resulting in an increase in woody vegetation. Woody plant encroachment in Mitchell grasslands in Australia, for example, has created situations where fire re-introduction may no longer be possible, thus causing shift in species composition (Burrows *et al.* 1986; Burrows *et al.* 1991; van Etten 2010). This is, according to Van Langevelde *et al.* (2016) is alternate vegetation state that can be irreversible (stable state). The concept of alternative stable state has been shown to be relevant when studying the dynamics of savanna ecosystems (Perrings and Walker 1997; Rietkerk *et al.* 2002).

1.4. PLANT RESPONSES TO FIRE

In savanna ecosystems, disturbance agents such as fire has been recognized as one of the key process that can affect diversity levels and change the community composition and structure (Hobbs and Huenneke 1992). Fire events could catastrophically change the ecosystem over a short time scale (Scheffer *et al.* 2001). However the absence of fire may also hinder reproduction of certain plant species thus reducing species diversity in a given community (Knuckey *et al.* 2016). Hence, these responses to fire initiate gradual readjustments in species interactions and ultimately composition through time in plant communities (Walker *et al.* 2007). Plant size also affects competitive hierarchies in savanna plant communities. In their juvenile stages, trees or woody species are inhibited from emerging by interaction with grasses and also fire, but once they have managed to overcome these obstacles and develop a canopy, trees can suppress grasses understorey and thereby will reduce burning frequency (Bond and Wilgen 1996; Skowno *et al.* 1999). When fire has been absent for longer intervals, savannas will converted to non-flammable secondary forests or thickets which become increasingly protected from further fire intrusion with age (Bond and Wilgen 1996; Rosleine and Suzuki 2013).

Vegetation responses to fire vary. Intense fire most likely kills or damages some of the plants, but others are more persistent and depend on fire for their regeneration (Pickett and White 1985; Riswan and Kartawinata 1991). Fire survival on plants primarily depends on the following traits namely: thick bark, crown architecture and resprouting from buds protected by bark or their subterranean position (Bond and Wilgen 1996; Marrinan *et al.* 2005). Certain woody plants, if the cambium is not damaged, can tolerate total defoliation by sprouting new shoots from epicormic buds, thus quickly regenerating the canopy. There are also plants with below-ground resprouting traits. In some woody plants, buds proliferate at the bases of stems and form a woody swelling known as a lignotuber or burl (Bond and Wilgen 1996). Resprouting is now recognized as a key functional trait among woody plants and as the basis for the persistence niche and Clarke *et al*., (2013) have developed the *BPR* (buds-protection-resources) scheme to understand resprouting in fire-prone ecosystems (Clarke *et al.* 2013). In support of this framework, there is an urgent need for studies on the floristic and geographic spread of resprouting trait types (apical, epicormic, basal, below-ground), which researchers hope will lead to a *pyrogeography* of fire responses across the globe (Clarke *et al.* 2013). In addition, there are also some species that characterize the fire-prone ecosystem that are not capable of resprouting and are killed by fire, and rely on mass regeneration after fire for persistence via seed stored at the site either in the topsoil or in woody fruits (obligate seeders). Meanwhile, other species are not killed by fire, but then die in the interval between fires. Nevertheless, their seed continues to exists in the community as dormant seed banks (above - or below-ground) and they may reappear after the next burn (Bond and Wilgen 1996). Therefore there is a need for an understanding of some key ecological traits for these savanna species present in Indonesia which will help determine the likely response and recovery of species following fire as such basic information is currently lacking. These traits include: depth from which the seed can emerge, longevity in the seed bank, seasonality of emergence, and germination triggers (James and Rahman 2013).

1.5. INVASIVE ALIEN SPECIES IN SAVANNA

One of woody species that is known to inhabit certain savanna ecosystems is *Acacia nilotica. Acacia nilotica* is known to be abundant in its native habitat in Africa (Brenan 1983), but has been scantly studied. In a study of savanna by Skowno (1999) in the Hluhluwe Game Reserve, South Africa, *A. nilotica* which is native to the area, was described in terms of its quantitative structure and distribution. In Australia, this species is dominant in Queensland where it has been declared as a noxious weed, and only a few are found in Western Australia, New South Wales, Adelaide and Northern Territory (DAFF 2014; Reynolds and Carter 1990). Adult trees of *A. nilotica* are fire tolerant and can have negative impact on savannas (Radford *et al.* 2001a). Impact of *A. nilotica* on Mitchell grasslands in Australia was studied by Burrows *et al.* (1991) where it was observed that *A. nilotica* was invading the grasslands and forming thorny thicket formations. Although adult trees of *A. nilotica* are apparently fire tolerant, the effects of fire on seed bank mortality, viability and germination are understudied. Similar phenomena can also be seen in Baluran savanna of East Java, Indonesia (Tjitrosoedirdjo 2008). *Acacia nilotica* was first introduced to Baluran National Park in the late 1960s, when its original purpose was to create fire breaks to prevent the spread of fire from Baluran Savanna into the adjacent teak forest. However, today, *A. nilotica* has spread rapidly and threatens the existence of Baluran savanna as it has been observed to cause changes from open savannas to more closed canopy in some areas (Barata 2000; Djufri 2004).

It should be noted that the currently accepted name according to International Plant Name Inex or IPNI (www.ipni.org) and The Plant List (www.theplantlist.org) for *Acacia nilotica* is *Vachellia nilotica*. However there is much controversy and debate regarding this change. As Kull and Rangan (2012) state in their thorough review: "The controversy over the resulting name changes reflects a range of concerns that some taxonomists regards as subjective and unscientific. These include aesthetics tastes, sensibility of place, territorial chauvinism and personal institutional politics". For Baluran National Park managers and field rangers, and surrounding locals, *Acacia nilotica* is the name that they are familiar with and have been using for quite a long time. They may find the name change is unnecessary or objectionable and may keep retaining the old name. For the purposes of this thesis, therefore, I will continue using the name *Acacia nilotica*. Tjitrosoedirdjo (2008) described *Acacia nilotica* found in Baluran as follows: "*Tree*, with an umbelliform crown, branching almost from the base, with a dark, fissured bark and a deep taproot, 5.00-10.00 m tall . Tops of branchlets are at first finely hairy, then soon glabrous. *Leaf* compound, bipinnate, a petiolar gland is present between the two lowest pairs of pinnae, pinnae 2-8 pairs, leaflets 12-28 pairs, linear, 2-5 mm long. *Head flowers* bright yellow, axillary or the higher ones racemed or panicled, 1-1.50 cm diam. *Fruits* are indehiscent and flat pods of 6-25 cm length and 1-1.50 cm width, containing 8-15 seeds of 6-7 mm diameter".

Over dominance of the woody species *A. nilotica* could shift the savanna into another ecosystem state, i.e. secondary forest. The widespread occurrence of *A. nilotica* in the Baluran savanna according to Tjitrosoedirdjo *et al*. (2013) may have been facilitated by large mammals such as the buffalo. At the end of the wet season, and into the dry season, mature *A. nilotica* pods drop from the trees and are consumed by herbivores. Tjitrosoedirdjo *et al*. (2013) suggested that the digestive system of the herbivores may scarify *A. nilotica* seeds, enhancing their germination, and that the herbivores would also facilitate the spread of *A. nilotica* in Baluran. However this suggestion is lacking in empirical evidence and specific research.

1.6. ROLE OF REMOTE SENSING FOR SAVANNA STUDIES IN INDONESIA

The role of remote sensing (RS) and geographical information systems (GIS) in ecology, especially in fire and vegetation management, has been recognized and indeed studies on mapping and analysing fire history are common (Arno *et al.* 1977; Chuvieco and Congalton 1989; Keane *et al.* 2001; van Wilgen *et al.* 2000; Verlinden and Laamanen 2006). A time series of fire scars was developed using all available Landsat images between 1989 and 2001 for an area of 63,000 sq. km in north-east Namibia to investigate the relationship between fire frequency, rainfall, and land cover (Verlinden and Laamanen 2006). Van Etten (1998) used a GIS for predictive vegetation mapping using models that linked vegetation units to mapped environmental variables across the extensive remote areas of Hamersley Ranges in Australia. This is one of the uses of remote sensing and GIS where conventional approach of vegetation analysis is difficult and expensive to conduct directly in the field (van Etten and Fox 2004; van Etten 1998). Furthermore, the development of remote sensing with hyper-spectral imaging was used by Schmidtlein *et al*., (2012) where he established a link between the three primary ecological strategies (or traits) and their canopy reflectance, and tested whether remote sensing is able to reproduce the spatial patterns of strategy types. His study demonstrates the potential to detect community strategy-type composition using hyper-spectral remote sensing, providing direct insights into spatial ecological patterns.

There has been considerable amount of studies on the use of RS/GIS in savanna ecosystems (Chacón-Moreno 2004; Hudak and Brockett 2004; Sano *et al.* 2010; Stroppiana *et al.* 2003), however, not many studies have addressed the use of RS/GIS in mapping forests-savanna boundaries (Banfai and Bowman 2005; Millington *et al.* 1992). One of the few studies was conducted by Millington et al., (1992), where they used advanced very high resolution radiometer (AVHRR) in mapping forests and savannas in sub-Saharan Africa.

Remote sensing and GIS have been also used in Indonesia and a range of literature has demonstrated the use of remote sensing and the available tools in GIS for various ecological studies such as for habitat suitability (Gamasari 2007), land use and cover change (Lavigne and Gunnell 2006), species conservation (Iskandar *et al.* 2012), invasive alien plant species (Setiabudi *et al.* 2013) and ecotourism (Untari 2009). Furthermore, fire mapping has also been undertaken (Fisher *et al.* 2006). However these studies are mostly concentrated on the issues of forest and peat land degradation, trans-boundary haze, and the impacts on biodiversity and economy associated with major El Nino Southern Oscillation (ENSO) events and volcanic eruptions (Fisher *et al.* 2006; Lavigne and Gunnell 2006; Siegert and Hoffmann 2000). Savanna fire has been particularly neglected, especially in the widespread savannas areas in East Nusa Tenggara (NTT). To date, no regional studies have been undertaken that reliably assess the seasonal extent and patterning of prescribed burning and wildfire. One of the few studies was done by Fisher *et al.* (2006) in Sumba Island East Nusa Tenggara. Their study adapted and applied standard remote sensing and geographic information system tools (as developed for monitoring fire patterns in savanna landscapes of northern Australia) to that of several locations in Sumba Island, East Nusa Tenggara, but only for two years. Remote sensing could provide insight on the patterns of habitat fragmentation in NTT ecosystems where savannas have the widest coverage (Tacconi and Ruchiat 2006).

1.7. STUDY RATIONALES AND AIMS

This study's broad objective is to improve our understanding of the vegetation characteristics, patterns and dynamics of savanna ecosystems in wetter parts of Indonesia Archipelago. Knowledge of the patterns and characteristics of savanna plants in Indonesia especially in the wetter regions is critical. Positioning Indonesia in the world's savanna literature may not be trivial however as the building blocks are yet to be established. The question of how Indonesian savannas fit with other classifications, models and theories on savanna, especially those developed for Africa and Australia, needs to be answered to fill the gap in the literature. Therefore this study is significant as fundamental building blocks for subsequent studies on Indonesian savannas.

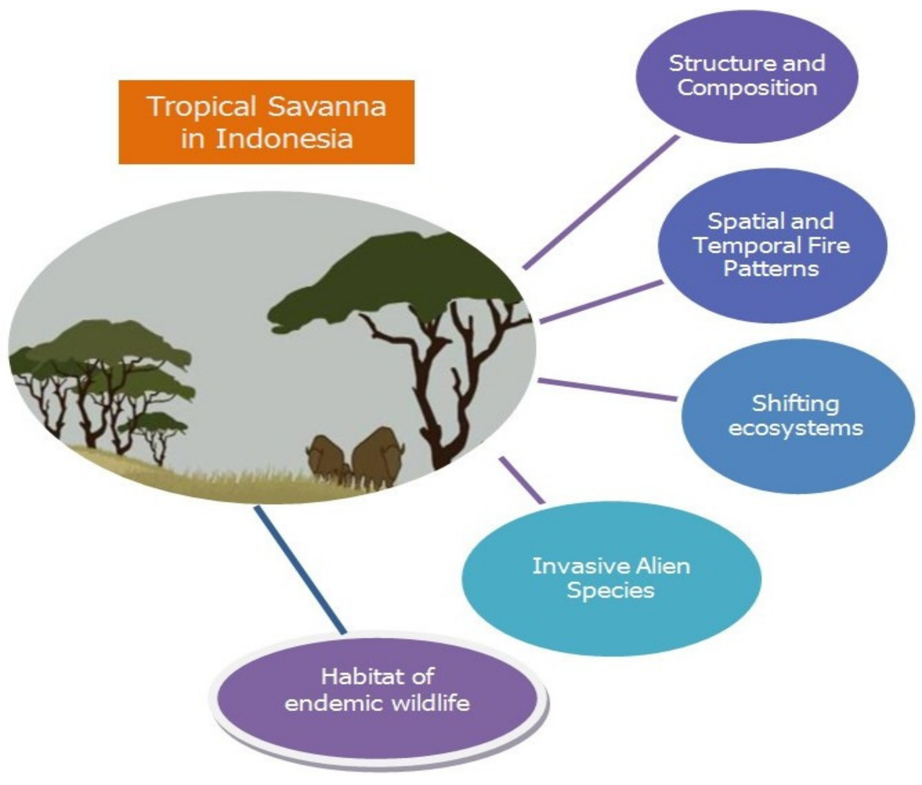


Figure 1. Conceptual diagram linking the various studies and chapters in this thesis

Given the framework (Figure 1) and background presented, the thesis is structured around five key objectives:

Objective 1: Establish characteristics of and variation in the structure, diversity and composition of savanna plant communities in the higher rainfall parts of Indonesia (Java, Bali, and Lombok Islands), and how they relate to environmental factors; also what do such characteristics tell us about the origin and age of these savannas (Chapter 2).

Objective 2: Describe spatial and temporal patterns of fire in savanna landscapes in the central Indonesian Archipelago (Chapter 3).

Objective 3: Characterize the structure and composition of seasonal dry tropical forest (SDTF) and savanna in Baluran National Park, East Java, Indonesia, and to evaluate the evidence for ecological transitions between SDTF and savanna (Chapter 4).

Objective 4: Assess the present issue of invasion in the context of conversion of savanna ecosystem to a homogenous stand of prickly acacia (*Acacia nilotica*) both spatially (using GIS) and via laboratory experiments, and to predict the potential spatial distribution of the species in Indonesia under global climate change (Chapter 5 and 6).

Objective 5: Characterize the dynamics of savanna and forest as important habitat for the endemic Bali Starling (*Leucopsar rothschildi*) in Bali Barat National Park (Chapter 7).

In the final chapter (Chapter 8), I integrate the findings of the main (research) chapters into a broad understanding of the role of fire and grazing in vegetation dynamics in a tropical savanna setting and how this information can help managers of the national parks. To this end, I also offer a new conceptual framework to view the interactions between rainfall, fire and grazing in establishing and maintaining savannas in Indonesia, particularly in the presence of prevalent invasive exotic species.

1.8. STUDY SITES DESCRIPTION

This study was conducted across numerous savanna sites and areas across the Indonesian archipelago from Java to Bali to Nusa Tenggara Islands via field visits and remote sensing. These savannas stretch from west to east, with precipitation being highest in the west and decreasing towards the eastern side of the Archipelago. Two savannas in Java, one savanna in Bali and one in Lombok (West Nusa Tenggara) were visited (Figure 2) namely: Baluran savanna, Alas Purwo savanna (East Java), Bali Barat savanna (Bali), and Gunung Rinjani savanna (Lombok). Soils in these savannas are of latosols type that was developed from Miocene sedimentary rock and volcanic rock as Indonesia is known as the "ring of fire" because of the occurrence of volcanoes that stretch across the Archipelago.

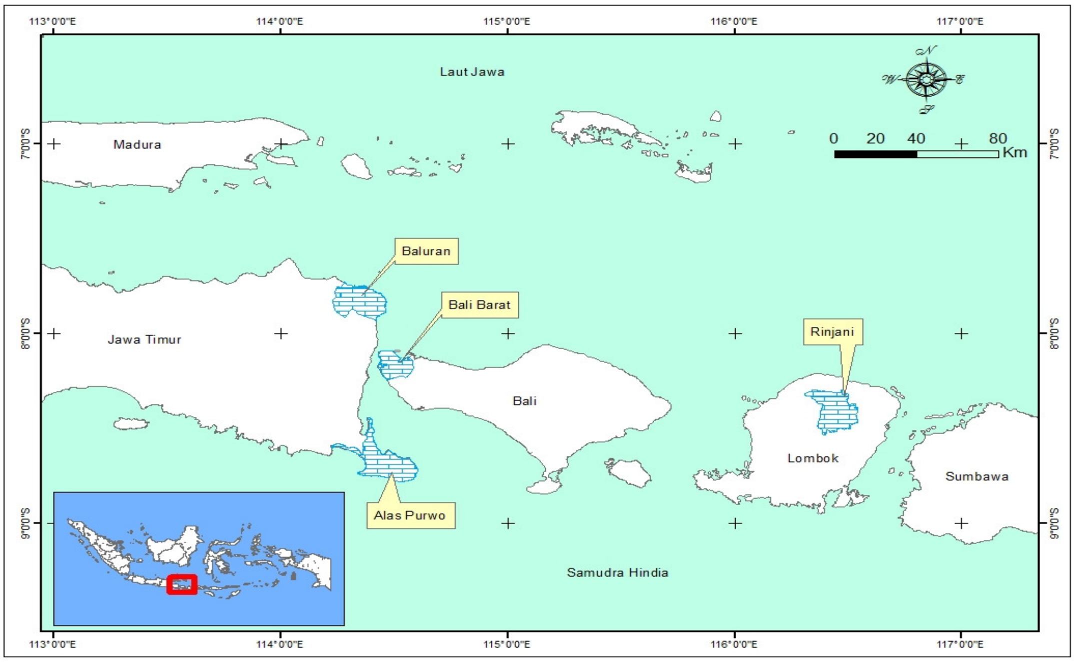


Figure 2. Locations of the study sites in Indonesia consists of two savannas in East Java (Baluran National Park and Alas Purwo National Park), one savanna in Bali (Bali Barat National Park) and one savanna in Lombok (Gunung Rinjani National Park),

Regarding fire policy or fire setting in these savannas, Ministry of forestry has applied zero burning regulation to all of national parks and other protected areas. Therefore in almost all of these savannas, intentional fire are not exist or very rare. Alas Purwo and Bali Barat Savanna for instance have experienced no fire for a significant time period. Baluran however, still conduct controlled fire management. This is because the park's management use fire as a mean to control the invasive alien species *Acacia nilotica* from spreading further in the areas, especially the Bekol Savanna. For savanna located in the Gunung Rinjani National Park, low intensity fire are still exist sporadically mostly because of burning of grasses by local people surrounding the national park that use the savanna as feeding ground for their cattle.

1.8.1. Baluran National Park Savanna

Baluran is located at the northern-most tip of East Java, on plains in the rain shadow of mountain ranges. Baluran National Park is located in Situbondo Regency, East Java Province; to the north it is bordered by the Madura Strait and on its east side it is bordered by the Bali Strait (Figure 2). The park covers an area of about 25,000 ha, located in between 7° 29' 10" and 7° 55' 55" South (latitude) and 114° 29' 20" and 114° 39'10" East (longitude), and is characterized by a single volcanic cone of Baluran mountain having a summit at 1,247 m (Tjitrosoedirdjo *et al.* 2013). In 1937 the Dutch Government proclaimed this area as a wildlife reserve (decree GB. No. 9 dated 25 September 1937 Stbl. 1937 No. 544) to conserve large mammals, mainly Javan *banteng* (*Bos javanicus* subsp. *javanicus*) that inhabit the surrounding areas. This decree was then reinstated by the Indonesian Agriculture Minister in 1962 (decree No. SK/II/1962 dated 11 May 1962) and then it proclaimed as a National Park in 1980.

Baluran has a relatively dry climate for Java and mainly consists of savanna (~40%) (Figure 3) as well as dry forests and mangrove forests. Baluran has a monsoonal season type affected by the dry easterly winds. According to the Schmidt and Ferguson (1951) classification, Baluran National Park has F and G (dry climate) with temperature ranging 27.2-30.9°C, average relative humidity of 77%, and average wind speed of 7 knots. The rainy season generally runs from November to April, and the dry season is from April to October. Highest precipitation is usually in December to January. Precipitation in Baluran National Park ranges from 900 - 1,600 mm/year. Baluran Mountain provides a rain shadow so that rainfall is lower and more seasonal than other parts of East Java. Baluran savanna has volcanic soils (vertisols). During the dry season or periods of drought, the soil will crack (which suggest clays or loamy clays) with the depth of the cracks reaching up to 80 cm.

The park authority divided the savanna areas in Baluran National Park by their topographic condition: flat savanna which covers an area of 1,500 – 2,000 ha (located in Bekol and Semiang) (Figure 3), and undulating savanna that covers an area of approximately 8,000 ha (located in Balanan, Kramat, Talpat, Labuhan Merak, Air Tawar and Karangtekok) (Sabarno 2002). The vast areas of savanna in Baluran National Park have led to its famous title as “a piece of African savanna in Java”.

Figure 3. Bekol Savanna in Baluran National Park, East Java (dry season 2014), seen from watch tower

Fire in Baluran has been a recurrent event. In the 1960s, fire was occurring naturally due to the dry season in Baluran. This reportedly caused the areas of the savanna to increase and the forested areas (especially the teak, *Tectona grandis,* forest stand) to contract. Therefore in 1980, the government introduced the exotic plant *Acacia nilotica* to the Baluran savanna in order to create fire breaks so that fire in the savanna did not spread into the adjacent teak forest stands (Sabarno 2002). However, it is now evident that this exotic species has now become invasive, not only causing a decrease in the area of the savanna, but also managing to invade the adjacent forest. In recent times, fires in Baluran have decreased from natural causes, and have become more anthropogenic. Whether its purpose is to eradicate the *Acacia nilotica* (this effort shows no substantial result as the species is somewhat fire resistant; (Djufri 2004), or it is due to poachers/illegal hunters or local people deliberately buring their lands, these human-mediated fires have spread into adjacent savannas (Sabarno 2002).

1.8.2. Alas Purwo National Park Savanna

Alas Purwo was originally named Banyuwangi Wildlife Reserve based on the Dutch Government Decree No. 6/Stbl. 456/1 September 1936. The wildlife reserve was then appointed as a National Park in 1992 based on the Indonesian Ministry of Forestry decree no. 283/Kpts-II/1992 and its name changed to Alas Purwo National Park. Alas Purwo National Park (APNP) is situated on the Blambangan Peninsula in Banyuwangi regency (8° 26’ 45” - 8° 47’ 00” LS and 114° 20’ 16” - 114° 36’00” BT), at the south-eastern tip of East Java province, along the shore the Bali Strait (Figure 2). With an area of 43,420 ha, the park is made up of mangroves, savanna, lowland monsoon forests and coral fringed beaches. Based on Schmidt-Fergusson climate classification, the APNP is classified as type D (relatively moist) to type E (relatively dry). Wet season usually lasts from November to April and dry season occurs in May through to October. The APNP receives an average of rainfall between 1,000 – 1,500 mm, with daily temperature average ranging from 25 - 28°C. The APNP also has little fluctuation in relative humidity across the year, ranging only from 75% to 81%. Alas Purwo National Park stretches from coastal areas, plains and lowland forest areas, up to hills forest with undulating topography at altitude ranging up to 322 m above sea level.

Figure 4. Sadengan feeding/grazing ground (Sadengan Savanna) in Alas Purwo National Park (dry season 2014), seen from watch tower

The main savanna area in Alas Purwo National Park is named the *Sadengan* Feeding Ground (SFG) (Figure 4). Originally, the SFG was open habitat in a lowland tropical forest of APNP where grasses and shrubs were dominant. The most common species were *Arudinella setosa*, *Alysicarpus vaginalis*, *Cyperus brevivolius*, *C. iria*, *Dischantium caricosum*, *Eleusine indica*, *Fimbristylis* sp., *Andropogon contortus*, *Ischaeum* sp., *Panicum repens*, *Paspalum conjugatum*, *P. vaginatum* and *Polytrias amaura*. In 1972, the National Park Authority decided to expand this area by clearing the surrounding, bordering forests to be then used as an area for herbivore feeding/grazing and wildlife conservation, mainly catering for large mammals such as Javan banteng (*B. j. javanicus*) as well as other species like *Cervus timorensis* and *Muntiacus muntjak*. Surveys of *Bos javanicus* have showed the decline in the number of individuals of this species (Hakim *et al.* 2005; Purnomo and Pudyatmoko 2011). One of the major causes is the decreasing area of the savanna (Hakim *et al.* 2005). Similar to other areas of Java, there is not comprehensive data on fire events in Alas Purwo National Park. There is only sporadic report and little information can be obtained (Mukti *et al.* 2016). However from the literatures, there were fire major events in APNP in 2003 (Artha 2011; Hakim *et al.* 2005; Reside *et al.* 2010).

1.8.3. Bali Barat National Park Savanna

On Bali Island, savanna can be found in the Bali Barat National Park (West Bali National Park/ BBNP). Bali Barat National Park is located on the north western side of Bali, Indonesia. The park area covers ~19,000 ha which constitute of terrestrial areas of 15,588 ha and marine areas that cover about 3,415 ha. This area of BBNP is approximately 5% of Bali's total land area. A seaport at Gilimanuk is located west of the park. The National Park can be reached by roads from Gilimanuk and Singaraja, or by using ferries from Ketapang, East Java. There are several distinct habitats in the national park: savanna, mangroves, montane and mixed-monsoon forests, and coral islands. In 1984, Bali Barat was declared as aNational Park based on Ministry of Forestry decree (No. 096/Kpts-II/1984). The national park has various topographic conditions ranging from plains to steep slope with an altitude up to 1,414 m above sea level. There are four mountains, namely Prapat Agung, Banyuwedang, Klatakan and Sangiang which is the highest one (at 1,414 m a.s.l). There are also four islands that are included the BBNP management areas, namely Menjangan, Burung, Gadung and Kalong Islands. Based on the Schmidt-Fergusson climate classification, BBNP has the D, E, and C type of climate with average rainfall ranging from 900 – 1,500 mm and average temperature of 33°C (Masy'ud *et al.* 2008; Masy'ud *et al.* 2007; Whitten *et al.* 1996). The national park also harbours more than 11 species of mammals, more than 160 species of birds and herpetofauna, and more than 120 species of coral reefs fish (Wiryawan 2014). But one species of bird of world reknown in Bali Barat National Park is the endemic bird the Bali Starling, often also called the Bali Mynah (*Leucopsar rothschildi*). This critically endangered bird species is the only endemic bird found in Bali. Stresemann (1912) collected and described the first Bali starling known to science near Bubunan, Bali. Bali Starling or Bali Mynah is an attractive aviary bird being largely white, with black wing and tail tips and with bare skin of a turquoise-blue colour on the lores and behind the eye.

Figure 5. Lontar *(Borassus Flabellifer)* savanna in Bali Barat National Park (dry season 2014)

In BBNP, savanna can be mainly found on the north side, in Tanjung Sari near the coastal areas. Savanna in BBNP (Figure 5) is characterized by the dominance of woody species *Acacia* spp., *Borassus flabellifer*, *Phyllanthus emblica* and also *Corypha utan*. On the ground layer, grasses, shrubs can be found, such as *Crotalaria* sp. and *Indigofera* sp. According to BBNP management (Wiryawan, 2014, personal communication), the savanna in BBNP is experiencing a decline in area due to invasion of exotic invasive shrubs species. Although savanna is not assigned as feeding ground, there are people from the surrounding area of BBNP that used this savanna illegally as grazing ground for their cattle (Sugiarto, 2014, personal communication). There is almost no literature available regarding fire in BBNP, however, Whitten *et al.,* (1996) wrote that around 1990, the understorey shrubs and below parts of the trunk from woody species dominant of displaying black colour, similar to fire scar, a clear sign that fire does occur in this savanna.

1.8.4. Gunung Rinjani National Park Savanna

Nusa Tenggara (Lesser Sunda Islands) stretches at the foremost south of the Wallaceae line and its biodiversity is mostly understudied (Monk *et al.* 2000; Raharjanngtrah and Rahman 2004). Nusa Tenggara is a group of small and medium islands stretches from Lombok Island in the west to Tanimbar Island in the east. These islands generally have low tree density with widespread grasslands, shrubland and savanna. According to Monk, et al., (2000), Nusa Tenggara Islands are dominated by mixed savanna (62%), lowland rain forest (19%), limestone forest (10%), sub-montane rain forest (5.5%), coastal forest and tidal, mangrove, montane forest, riparian and ultra basic form the rest of the coverage on the islands.

Gunung Rinjani National Park (GRNP) is located on Lombok Island, West Nusa Tenggara, at 116°21’30” - 116°34’15” E and 8°18’18” - 8°32’19” S. Initially GRNP was a wildlife reserve which was proclaimed by the Dutch government in 1941 (Staatblaat no. 77/12 March 1941). It retained this status until in 1997 when the Indonesian Forestry Ministry appointed Gunung Rinjani as a National Park which covers vast areas of approximately 41,330 ha. Gunung Rinjani National Park stretches from an altitudinal range of 500 – 3,726 m asl (summit of Mount Rinjani) and has topographical feature of plain, undulating, and steep slope landscapes. Gunung Rinjani National Park receives average rainfall of ~2,000 mm per year and average daily maximum temperatures ranges from 23° to 30°C at 550 m asl. The forest has been classified into three vegetation zones according to elevation: lower montane (600–1,500 m asl), pre-montane (1,500–2,000 m asl) and montane (2,000–2,600 m asl) (Dossa *et al.* 2013). Rinjani lies within the major transition zone of Wallacea and therefore its flora and fauna shows a dramatic transition from South East Asian species into those which are typical of Australia and New Guinea.

Figure 6. Savanna on the slope of Mount Rinjani, in Sembalun Village, Gunung Rinjani National Park (dry season 2014)

Rinjani National Park consists of 40% primary forests, 40% of savanna (Figure 6) and 10% of planted forest. The GRNP protects several endangered plants such as, *Pterospermum javanicum*, *Swietenia macrophylla*, *Ficus superba*, and *Toona sureni* to name but a few, as well as animals such as *Presbytis* sp., *Philemon buceroides*, and *Lichmera lombokia* (scaly-crowned honeyeater) as the flagship species (Raharjanngtrah and Rahman 2004). Savanna in GRNP can be found in Sembalun area (Figure 6). Sembalun is one of two hiking tracks to Mount Rinjani. This track is one of the favourites of mountain climbers because it is not too steep. *Imperata* as well as other grass species occur together with woody species such as *Casuarina* and *Albizia chinensis* (Monk *et al.* 2000) . In 2009, GRNP reported that 300 ha of savanna on the south slope of Mount Barujari (500 m from Segara Anakan Lake) were burnt. Rinjani authorities believed that it was caused by poachers or local people that deliberately lit the fire (Khafid 2009). Mount Rinjani is undergoing primary succession in most parts of its south-west slope following an eruption in 2009 (ESDM 2014b). However once grasses recolonised these slopes, only small amount of tree species have grown. Clumps of trees formed a small “island” of forest found in the crevices and steep gullies (Figure 6).

Chapter 2: Savanna Plant Communities in the Wetter Parts of the Indonesian Archipelago

ABSTRACT

Little is known about tropical savanna communities in South East Asia. In fact, the description of savanna in the Indonesian Archipelago is scarce in this region which, based on climate alone, would be expected to be forest. I compare and contrast the vegetation characteristics of four savannas in the wetter part of the Indonesian archipelago (Java – Bali – Lombok) to address two important questions: 1) what environmental factors are linked to, and therefore may be driving, differences in savanna floristic composition; and 2) what do these vegetation features tell us about the origin, maintenance and age of these savannas? I hypothesized that floristic gradients across these savannas would be primarily linked to elevation, precipitation and related micro-climate differences, with disturbance type and frequency being of secondary importance. Field work was conducted in eastern Java (Baluran National Park & Alas Purwo National Park), Bali (Bali Barat National Park) and Lombok (Rinjani National Park), Indonesia. The study areas were cross-checked with a fire map based on MODIS burn area from year 2000 to 2013 to obtain typical fire regime of the savannas. Field sampling was conducted in 2014, using plots to record abundance and cover of savanna plant species (trees and groundcovers layers). Local environmental data for every plot were also recorded at a similar time of day. I used Importance Value Index (IVI), Shannon-Wiener species diversity index and Margalef species richness to analyse the structure-composition and diversity of the savannas. Variation in community composition was evaluated using analysis of similarity (ANOSIM) and correlation with environmental factors was explored using canonical correspondence analysis (CCA). Overall my study showed there were distinct gradients in elevation (along with related climatic factors such as temperature and precipitation) and fire regime linked to floristic composition across the savannas of Java, Bali and Lombok Islands. Each savanna was characterized by different set of woody and grass species, with Invasive Alien Species (IAS), such as *Acacia nilotica, Lantana camara* and *Chromolaena odorata,* being particularly important in differentiating between savannas. Characteristics of species from the Baluran savanna suggest this ecosystem may be of considerable age, whereas the other savannas are likely to be maintained by regular fire. This study is the first study to describe more thoroughly the savanna plant community in the wetter parts of Indonesian archipelago. Knowledge gained will be valuable as a building block for further studies on Indonesian savanna and may also be relevant to other tropical savannas of Southeast Asia.

Keywords: Savanna, Indonesia archipelago, plant community, gradient analysis, invasive alien species

2.1. INTRODUCTION

The remarkably diverse tropical forest communities of Indonesia have been widely recognized for their importance to the world's biodiversity and ecosystem services (Blackie *et al.* 2014; Dossa *et al.* 2013; Riswan and Kartawinata 1991). The structural and floristic variation of these Indonesian tropical forests have been extensively studied and related to variation in temperature, precipitation, seasonality of rainfall, edaphic conditions, topography, natural and anthropogenic disturbance, altitude, climate change and invasive alien species (Brearley *et al.* 2004; Harger 1995; Sutomo *et al.* 2015; Tjitrossemito 2012; van der Kaars and Dam 1997; Widyatmoko and Burgman 2006). However, little is known about tropical savanna communities in Southeast Asia (Bond and Wilgen 1996; Furley 2004; Werner 1991). In fact, the description and study of savanna across the Indonesian Archipelago is scarce. Savannas are tropical grass-dominated ecosystems with sparse to mid-dense woody plants, but throughout much of Indonesia, such vegetation would be, based on climate alone, expected to be forest (Bond and Keeley 2005).

Savannas in Indonesia are found across the archipelago over a range of climate and soils (Monk *et al.* 2000; Whitten *et al.* 1996). The most well-known and studied occur on the islands of East Nusa Tenggara (Indonesian: Nusa Tenggara Timur; NTT) in the driest eastern parts of the archipelago which have pronounced seasonal rainfall, such as West Timor, Sumba and Flores (Fisher *et al.* 2006; Monk *et al.* 2000; Russell-Smith and Edwards 2006; Tacconi and Ruchiat 2006). Species composition of savanna in NTT was studied by Auffenberg (1981) and described in terms of dominant woody species. *Borassus flabellifer* (Arecaceae/Palmae) dominated the tree layer of savanna at Komodo Island, Rinca Island and the northern and southern coast of Flores Island up to an altitude of about 400 m above sea level (asl), whereas *Ziziphus mauritiana* was found growing from sea level up until 500 m asl. Other types of savanna are also commonly found in the eastern part of Indonesia (based on major tree species), namely *Eucalyptus* savanna in Timor, and *Casuarina* savanna in Sumba and Timor (Goltenboth *et al.* 2006). However, detailed information on the occurrence and features of savanna in Indonesia, especially in the wetter regions, such as Java, Bali and Lombok, is scarce. Whitten *et al*. (1996) mention the existence of savanna in Baluran, East Java and Bali Barat in Bali, however they did not discuss this thoroughly.

Savanna ecosystems in Southeast Asia have long been regarded as anthropogenic, being derived from tropical forests and maintained via ongoing human manipulation, primarily clearing, grazing and/or burning (Ratnam *et al.* 2011; Solbrig *et al.* 1996; Stott 1990). This view has certainly been widely reported for Indonesian savannas (Goltenboth *et al.* 2006; Whitten *et al.* 1996). However a recent review by Ratnam *et al*. (2016) questions this assumption and points to antiquity of some Southeast Asian savanna. Evidence for this comes from: 1) fossil history and phylogenetic data showing existence of savanna species of plants and animals in the region before humans; 2) dominance by species with adaptations to withstand repeated fire and/or grazing; and 3) climatic consistencies with savannas of other continents.

In this chapter, I compare and contrast the vegetation characteristics of four savannas in the wetter part of the Indonesian archipelago (Java – Bali – Lombok) to address two important questions: 1) what environmental factors are linked to, and therefore may be driving, differences in floristic composition across these savannas; and 2) what do these vegetation features tell us about the origin, maintenance and age of these savannas? I hypothesized that floristic gradients across these savannas would be primarily linked to elevation, precipitation and related micro-climate differences, with disturbance type and frequency being of secondary importance.

2.2. METHOD

2.2.1. Study Sites

The study was conducted in four Indonesian savannas on Java, Lombok and Bali Islands. These were selected based on all the well-known localities of savanna ecosystems in this region as reported in the literature (Whitten *et al.* 1996). Two savannas were located in East Java (Baluran National Park and Alas Purwo National Park), one in Bali (Bali Barat National Park) and one in Lombok (Rinjani National Park) (Figure 1). Other savannas reported in the literature were visited, but not studied here as no longer savanna (e.g. Pangandaran Nature Reserve in West Java, now largely converted to secondary forest). Typical fire regimes and climate for each savanna studied are described in Table 1. In all of these savannas, the soils are of volcanic origin, but differing in age.

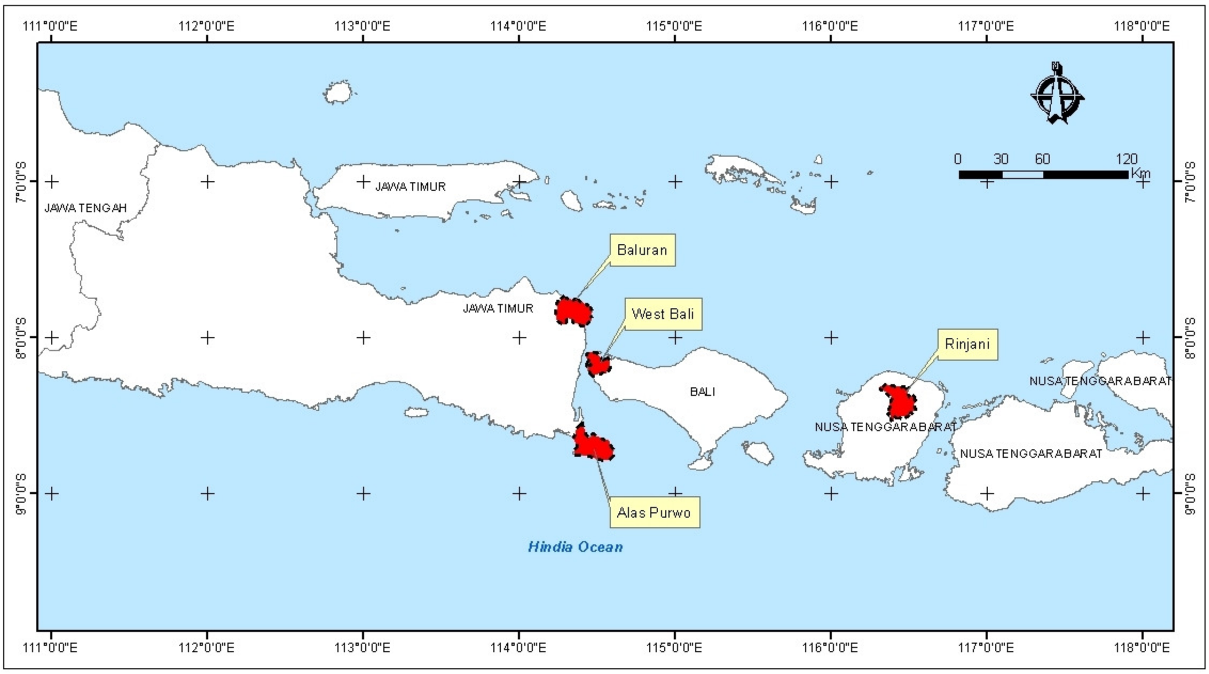


Figure 1. Locations of the national parks across Java, Bali and Lombok Islands Indonesia

Table 1. Typical fire regime, average precipitation and average temperature

Savanna site	Fire frequency (per decade)	Time Since Last Fire	Average Precipitation (mm/year)	Average Temperature (°C)
Alas Purwo National Park	1	~ 10 years	1079 - 1554	26 – 29
Rinjani National Park	2	~ 6 years	2000	23 - 30
Bali Barat National Park	1	~ 10 years	972 - 1550	27 - 29
Baluran National Park	4	~ 4 years	900 -1600	27-37

2.2.2. Field Sampling

Between September to November 2014 (dry season), ten sampling plots (50 x 50 m) were randomly selected and established in each savanna area (Java, Bali and Lombok Islands; 40 plots in total); they were spaced at least 200 m apart. Inside each of the 50 x 50 m plots, I had four smaller plots of 5 x 5 m nested randomly. Inside the 50 x 50 m plot I identified, measured and recorded all tree species ≥ 10 cm diameter at 1.3.m (dbh). In the smaller-nested plots I identified all groundcover species (grasses, ferns andforbs) and estimated their cover. Plant identification was primarily done using the resources of the Hortus Botanicus Baliensis, Bali Botanical Garden, but also the Flora Malesiana (http://floramalesiana.org/), PROSEA (Plant Resources of South East Asia; http://proseanet.org) and the Bioportal at Naturalis Biodiversity Centre, the Netherlands (http://bioportal.naturalis.nl). Nomenclature follows the Plant List (http://www.theplantlist.org/) maintained by the Royal Botanic Gardens, Kew and Missouri Botanical Garden. I also recorded local environmental data for every plot, at a similar time of day (morning): soil pH and moisture, local microclimate (light intensity, air temperature, relative humidity, and wind velocity and heat stress index) and topography (altitude and slope). The plot locations were cross-checked with fire scar maps produced using the MODIS burned-area product (Roy *et al.* 2008) for the years

2000 to 2013 to obtain information on time since fire and fire frequency. Typical fire regime of the savanna areas was obtained from studying this fire map of the regions, complemented by interviews with park rangers and field observations.

2.2.3. Data Analysis

The Importance Value Index or IVI (Kent 2011) was calculated for each species in each plot and then averaged to understand the floristic dominance and plant community composition in each savanna. Species richness and the Shannon-Wiener species diversity and evenness indices (Magurran 2004) were also calculated for each plot. Differences in mean values between savannas were tested for significance using ANOVA and Tukey's b post-hoc tests in SPSS (v.23, IBM Corp., New York, U.S.). The species cover data were square-root transformed prior to constructing a resemblance matrix based on Bray-Curtis similarity (Valessini 2009). A Non-metric Multidimensional Scaling (NMDS) ordination diagram was first generated based on the resemblance matrix. The compositional differences between savannas were then tested for significance using one-way ANOSIM (Analysis of Similarity) (Clarke 1993). SIMPER (Similarity Percentage) analysis was then used to explore the relative contribution of individual species to the total dissimilarity among savannas. These multivariate analyses were performed using PRIMER (version 6.0., PRIMER-E Ltd., Plymouth, U.K.). Correlations between floristic and local environmental gradients was explored using the BEST (Bio-Env+Stepwise) method (Clarke and Ainsworth 1993) in PRIMER, as well as Canonical Corespondence Analysis (CCA) using CANOCO (version 5, Microcomputer power, Ithaca, New York).

2.3. RESULTS

2.3.1. Structure and Dominant Species

I discovered as many as 43 plant species within 26 families across the four savannas including one fern, seven grass or grass-like plants and two forbs (a full list of species, habit and their typical habitats are presented in Appendix 1). Each savanna has structural characteristics and dominant species that differentiated it from the others (Table 2). For instance, Alas Purwo is dominated by the grass *Arundinella setosa* and small shrubs *Desmodium laxiflorum* (Fabaceae) and the invasive alien species (IAS) *Chromolaena odorata* (Asteraceae) in the groundcover layer, whereas the tree layer is occupied by *Flacourtia rukam* (Salicaceae). At Rinjani, the groundcover layer is dominated by the grass *Imperata cylindrica* and the fern *Gleichenia microphylla,* whereas the tree layer is mainly composed of *Engelhardia spicata* (Juglandaceae). Bali Barat groundcover layer is also dominated by the Poaceae family (*Calamagrostis australis*) and the tree layer mainly consists of *Borassus flabellifer* (Arecaceae). In Baluran, the groundcover layer is characterized by *Desmodium laxifolium* (Fabaceae), seedlings of *Azadirachta indica* (Meliaceae) and two grasses *Polytrias indica* and *Dichanthium caricosum,* whereas the tree layer mainly consists of species such as *Ziziphus mauritiana* (Rhamnaceae) and the IAS *Acacia nilotica* (Fabaceae). The presence of invasive alien species is notable in most of the savannas (Appendix 1) and influenced their structure and composition. Bali Barat and Alas Purwo face similar problems from the obnoxious invasive alien species *Chromolaena odorata*, while at Baluran, *Acacia nilotica* occurs widely on the ground (i.e. seedlings and small saplings) and in the tree layers (Table2).

Table 2. Importance Value Index (IVI) and cover (%) for the dominant and common groundcovers and trees species (IVI score higher than 10) from each savanna sampled. Asterisks indicate likely exotic species (see Appendix 1).

Species	**Family**	**IVI**	**Cover (%)**
	Groundcover/understorey layer		
Alas Purwo Savanna			
Arundinella setosa	Poaceae	93.6	65.9
Desmodium laxiflorum	Fabaceae	42.1	19.8
*Chromolaena odorata**	Asteraceae	30.1	5.9
Cyperus brevifolius	Cyperaceae	10.8	5.3
Flacourtia rukam (seedling)	Salicaceae	10.4	2.1
Mt. Rinjani Savanna			
Imperata cylindrica	Poaceae	48.3	27.2
Gleichenia microphylla	Gleicheniaceae	21.8	5.2
Melastoma polyanthum	Melastomataceae	14.9	4.3
Bali Barat Savanna			
Calamagrostis australis	Poaceae	100	78.4
*Lantana camara**	Verbenaceae	21.2	3.6
*Chromolaena odorata**	Asteraceae	18.5	3.5
Borassus flabellifer (saplings)	Arecaceae	15.2	2.7
Baluran Savanna			
Azadirachta indica	Meliaceae	61.4	39.2
Desmodium laxiflorum	Fabaceae	47.7	36.6
Polytrias amaura	Poaceae	29.7	7.6
Dichanthium caricosum	Poaceae	19.7	8.6
*Acacia nilotica**(seedlings)	Fabaceae	13.7	2.6
	Tree Layer		
Alas Purwo Savanna			
Flacourtia rukam	Salicaceae	79.1	50.6
Streblus asper	Moraceae	60.5	38.3
Mt. Rinjani Savanna			
Engelhardia spicata	Juglandaceae	183.9	35.3
Ficus septica	Moraceae	35.5	6.7
Laportea stimulans	Urticaceae	20.7	6.7
Bali Barat Savanna			
Borassus flabellifer	Arecaceae	75.1	38.2
Bridelia stipularis	Phyllanthaceae	50.4	29.4
Albizia chinensis	Fabaceae	25.2	14.7

Baluran Savanna			
Ziziphus mauritiana	Rhamnaceae	161.7	42.9
*Azadirachta indica**	Meliaceae	94.7	41.8
*Acacia nilotica**	Fabaceae	43.5	14.3

2.3.2. Diversity Measures

There are significant differences among the savannas in terms of Shannon-Wiener species diversity and species richness (Table 3). Rinjani has the highest species diversity compared to the others, while Alas Purwo has the lowest. There are no differences in species diversity between Baluran and Bali Barat. Significant differences were detected in the mean species richness of savannas, namely between Alas Purwo and Baluran, and between Rinjani and Bali Barat (Table 3). Bali Barat retains the highest species richness compared to others and Alas Purwo remains the lowest in terms of species richness, as well as species diversity. In terms of species evenness, Baluran has the highest evenness score, whereas Bali Barat has the lowest (Table 3).

Table 3. Differences in mean species diversity, species richness and evenness (per 400 m^2) between savannas. Different superscript letters (a, b, c) after mean values (±SD) indicate significant differences between sites as assessed with Tukey's b test.

Savanna	Shannon-Wiener Species diversity	Species Richness	Species Evenness
Alas Purwo	0.61 (± 0.30)^{a}	2.60 (±0.52)^{a}	0.61 (±0.27)^{b}
Rinjani	1.07 (± 0.45)^{b}	4.20 (±1.87)^{b}	0.64 (±0.23)^{ab}
Bali Barat	0.72 (± 0.25)^{c}	5.00 (±1.05)^{b}	0.43 (±0.11)^{a}
Baluran	0.74 (± 0.40)^{c}	3.10 (±1.20)^{ab}	0.83 (±0.09)^{c}

2.3.3. Differences in Community Composition

The global test of analysis of similarity (ANOSIM) showed there were significant differences in Bray-Curtis species similarity between the savanna sites (Global R = 0.94; P<0.001). There is also clear separation of savanna sites in the ordination (Figure 2). Baluran savanna had the lowest value of average similarity (48.1%), indicating it had the greatest variation in community composition between the plots. The reverse is true for Alas Purwo (Figure 2). SIMPER analysis showed that twenty two species were mostly responsible for the dissimilarity between the sites (Table 4). From all six combinations of pair-wise comparisons of savanna areas, eleven species were important species in three savanna or more pair-wise comparisons (*Acacia nilotica, Albizia chinensis*, *Borassus flabellifer, Chromolaena odorata, Engelhardia spicata*, *Flacourtia rukam*, *Gleichenia microphylla*, *Imperata cylindrica*, *Melastoma polyanthum Polytrias indica*, *Thespesia lampas*), and six species (*Antidesma bunius*, *Calamagrostis australis*, *Lantana camara*, *Ocimum tenuiflorum*, *Passiflora foetida*, *Streblus asper*) occurred in two savanna pair-wise comparisons. The species that contribute most to the dissimilarities among almost all pairs of savanna was *Desmodium laxiflorum* (Table 4). This species was present in all three savanna sites (Baluran, Alas Purwo and Rinjani), but not in Bali Barat savanna.

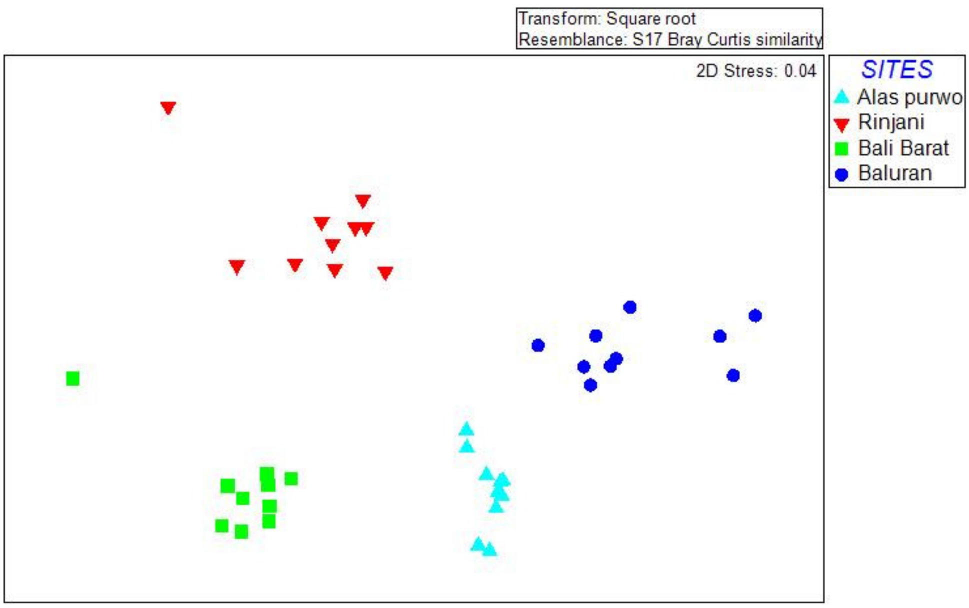


Figure 2. Result from NMDS Ordination (square root transformation, Bray-Curtis similarity) of vegetation data in four savannas in Indonesia. ANOSIM Global R statistic = 0.94 (P<0.001). Average similarity within savanna: Baluran = 48.1, Alas Purwo = 64.7, Bali Barat = 55.12, & Rinjani = 50.03.

Table 4. SIMPER analysis results showing percentage (%) contributions of species to average Bray–Curtis dissimilarities between pairs of savanna areas. Abbreviation refers to Baluran NP (BA), Alas Purwo NP (AP), Bali NP (BB), and Rinjani NP (GA).

Savanna sites comparison							
Species	**Family**	**AP & GR**	**AP & BB**	**GR & BB**	**AP & BA**	**GR & BA**	**BB & BA**
Average dissimilarity (%)		98.6	95.4	98.5	85.4	98.7	99.9
Acacia nilotica	Fabaceae	-	-	-	2.83	2.81	2.29
Albizia chinensis	Fabaceae	-	1.23	1.33	-	-	1.35
Antidesma bunius	Euphorbiaceae	-	2.88	-	-	-	3.14
Borassus flabellifer	Arecaceae	-	5.44	5.84	-	-	5.94
Bridelia stipularis	Phyllanthaceae	-	3.22	3.43	-	-	3.49
Chromolaena odorata	Asteraceae	-	5.33	4.09	-	-	4.17
Desmodium laxiflorum	Fabaceae	12.95	11.89	-	10.63	11.27	9.47
Calamagrostis australis	Poaceae	-	26.22	28.12	-	-	28.63
Engelhardia spicata	Juglandaceae	3.49	-	3.31	-	4.11	-
Flacourtia rukam	Salicaceae	2.33	2.07	-	2.78	-	-
Gleichenia microphylla	Gleicheniaceae	7.3	-	6.95	-	8.47	-
Imperata cylindrica	Poaceae	20.35	-	19.36	-	23.67	-
Lantana camara	Asteraceae	-	4.73	-	-	-	5.17
Melastoma polyanthum	Melastomataceae	4.92	-	4.66	-	5.82	-
Ocimum tenuiflorum	Lamiaceae	3.05	-	2.92	-	3.49	-
Passiflora foetida	Passifloraceae	-	-	-	5.49	-	4.45
Polytrias indica	Poaceae	-	-	-	11.87	5.45	9.62
Streblus asper	Moraceae	-	1.13	-	1.54	-	-
Thespesia lampas	Malvaceae	-	-	-	6.03	5.95	4.88

2.3.4. Savannas Community Composition Correlation with Physical-Environmental Factors

Canonical Correspondence Analysis (CCA) (Figure 3) shows that Rinjani (GR) in Lombok separates from the other savannas due to altitude and associated climatic differences, whereas Baluran (BA) is clearly separated by fire (higher frequency and less time since last fire) and, less so, by greater slope. In general, BEST (Bio-Env) analysis shows that altitude and precipitation are factors with the highest correlation to species composition across all four of sampled savannas (BEST Global test sample statistic Rho = 0.6; p =0.001).

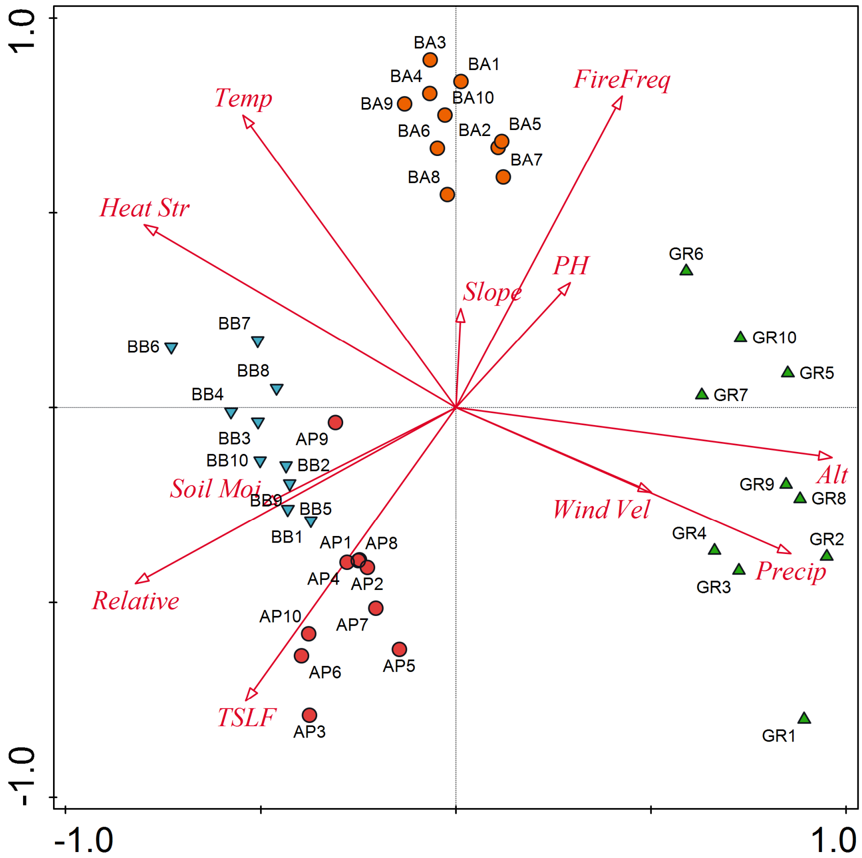


Figure 3. Results from Canonical Correspondence Analysis (CCA) showing a biplot of sample sites and environmental variables. Temp= air temperature; Heat Str= heat stress index; Relative= relative humidity; Precip= precipitation; Alt= altitude; Ph= soil pH; Slope= slope Wind vel= wind velocity; Soil moist= soil moisture; FireFreq= fire frequency; TSLF= time since last fire. Baluran NP (BA), Alas Purwo NP (AP), Bali NP (BB), and Rinjani NP (GA). PC1 and PC2 make up 63.2% of cumulative percentage of variation across samples.

2.4. DISCUSSION

In this study I have characterised the tropical savanna plant communities of the wetter regions of Indonesia distributed across three main Islands, namely Java, Bali and Lombok and their relationships with the physical environment. I found each of the four savanna ecosystems were distinctive in terms of species composition, types of species, physical environment and physiognomy. I confirmed my main hypothesis that climate and altitude differences are likely to be the main drivers for savanna plant community differences. However fire also potentially plays an important role as I found fire regime gradients were associated with the differences in community composition between the savannas. However I recognise that fire and climatic variables were derived from coarsely mapped data and, consequently, varied little between plots within each savanna. Therefore, the environmental gradients recognised are more likely to explain floristic dissimilarities between savannas, and more work is required to characterise floristic - environment relationships within discrete areas of savanna.

Savanna sites have a defining feature of domination of groundcovers, especially grasses and forbs, with sparse tree cover (Frost *et al.* 1986). All four savanna sites in my study fall in this criteria but each one of them is characterised by different set of tree species. The Bali Barat is characterised by palm species (*Borassus flabellifer*) as the tree layer. The same species also characterised the savanna in the drier part of Indonesian archipelago, such as in Kupang, Timor Island in East Nusa Tenggara (Monk *et al.* 2000). Species of the same genus (*Borassus aethiopum*) also characterise the humid Lamto savanna of the Cote d'Ivoire in West Africa (Barot *et al.* 1999) and indeed palm savannas are known in Venezuela (Holbrook and Putz 1996), Namibia (Konstant *et al.* 1995), as well as islands such as Madagascar and Mauritius (Safford 1997). In Baluran National Park (but not in area sampled), near Bama Beach, one species of palm (*Corypha utan*) is known to occur. These palms in Bali Barat, Kupang and Baluran occur in lowland areas, especially near to the coast. Some palms appear to be adapted to specific edaphic conditions, especially those related to drainage and soil depth (Widyatmoko and Burgman 2006).

In the Baluran savanna, common species in the tree layer are *Ziziphus mauritiana, Azadirachta indica*and *Acacia nilotica*. The same species has been reported from the drier eastern parts of Indonesia, such as Komodo, Rinca and Flores Islands, East Nusa Tenggara, from sea level up until 500 m asl. *Z. mauritiana* also occur in savanna in India (Pandey and Singh 1991). The same genus (*Ziziphus*) is common to many savannas of Africa, Arabia and south/SE Asia (Hess *et al.* 1996; Ratnam *et al.* 2016). In fact, of the four savannas studied, Baluran features typical savanna-type tree species showing adaptations to herbivory (thorns) and drought (small leaves, open architecture) and suggest it may be a relatively old savanna (*sensu* Ratnam *et al*. 2016). Even the invasive species *Acacia nilotica* and *Azadirachta indica* are typical savanna species (Dhileepan 2009; Radford *et al.* 2001b; Swaine *et al.* 1992). It is not clear if *Z. mauritiana* (Indian jujube) is exotic or native to Baluran as the species is widely cultivated across Asia (for food and medical products), but has uncertain origin and much disputed taxonomy (Islam and Simmons 2006; Janick and Paull 2008). Other evidence of the relative antiquity of the Baluran savanna are the dominance of C4 grasses, its open structure and persistence of native forbs, as well as the continued presence of indigenous grazing ungulates (e.g. *Bos javanicus*). The savanna here occurs in a rain shadow of the Mt. Baluran and appears to have lower and more strongly seasonal rainfall compared with the other savannas studied.

In terms of groundcover, savanna included in my study share similar common species with those found in other savannas of Southeast Asia, especially in Thailand, where species such as *Imperata cylindrica*, *Lantana camara and Chromolaena odorata* are all common place (Kodandapani 2013; Kurz 1876; Ratnam *et al.* 2016). C4 grasses (mostly from the Andropogineae clade) dominate the savannas, as expected for savannas, although *Calamagrostis australis*, which is widespread in the Bali Barat savanna, appears to be a C3 grass based on studies of other species in the genus (Osborne *et al.* 2014; Appendix 1). Another species which prominently contributes to the compositional similarities between my savannas is the tropical herbaceous legume *Desmodium laxiflorum. Desmodium* spp. are native to tropical Southeast Asia and Pacific Islands (Lenne 1981; Woomer *et al.* 1988) and are known to occur in higher rainfall savannas where they can survive fire and are known to be a palatable species to grazing animals. The species is absent in the Bali Barat savanna, whereas it is present in the other savannas studied. *Desmodium* perhaps requires regular ground disturbance, such as fires, to persist, something which has been largely excluded as part of the management implemented at Bali Barat National Park in recent times.

Although there are a few species in common across the savannas, there is very high floristic dissimilarity between them, especially in the tree layer. At the family level they are more similar, with certain families common to most (e.g. Fabaceae). There are several possible explanatory factors for the dissimilarity, namely geographic, geologic and climatic features, as well as management including use of fire. Baluran in East Java and Bali Barat in Bali for example, are both national parks and are relatively close to each other (~60 km,) but they are separated by Bali Strait and are climatically different with Baluran having lower and more strongly seasonal precipitation and higher temperatures. This leads to more fire-prone vegetation at Baluran, which also is regularly prescribed to control the domination of woody plant, particularly the invasive alien species *Acacia nilotica* in the savanna, whereas in Bali Barat, fire is now suppressed and the savanna is shifting to dry forest or secondary regrowth (Sutomo & Van Etten, unpublished paper). Also there were many invasive woody plants and climbing plants in the Bali Barat savanna, which is likely to reflect its proximity to human settlements and activity. Similarly at Alas Purwo, domination by many invasive alien species, especially *Chromolaena odorata*, was recorded (Appendix 1). Both Bali Barat and Alas Purwo savannas are in danger of being transformed into a forest structure due to lack of fires and prevalence of invasive species.

Globally, savannas located in high rainfall areas typically have at least some woody vegetation and in the absence fire can have high tree cover that would outcompete and suppress grasses species, thereby disenabling further fires and facilitating a transition to a forested ecosystem (Rosleine and Suzuki 2013; Scholes and Walker 1993). Thus, regular fire is important to establish grass-tree coexistence in these areas (Bond and Keeley 2005; Sankaran *et al.* 2003). Sumardja and Kartawinata (1977) described savanna in Pangandaran Peninsula, West Java and reported that some savanna sites had been abandoned around 1957 and have not experienced any fire since then; subsequently, they have become young secondary forests dominated by *Decaspermum fruticosum*. In contrast, savannas of drier sites, woody vegetation is mostly in low abundance whereas grasses are the dominant component. Semi-arid savannas have resource limitations (mostly competition for water) and lower biomass/fuels, and therefore fire is perhaps less influential for grass-tree coexistence in this type of savanna (Sankaran *et al.* 2005). On Java Island, especially in East Java, recolonisation (primary succession) after lava flow and volcanic eruptions is initially dominated by grasses which may then be maintained by regular fire over long periods. Therefore in this region of Java, which is subject to a more pronounced dry season (less rain and longer period of dry season) than central and western parts of the island, fire is believed to be the major factor that created and then maintained savannas (van Steenis 1972).

Rinjani on the island of Lombok is also very distinct. This savanna is located at higher elevations on the slopes of the volcano Mt. Rinjani and has been maintained in savanna form presumably since the early stages of primary succession after the last major eruption and lava flows in the area of study. Once grasses recolonised and dominated the area, only a small number of tree species have developed, perhaps due to high competition for resources with grasses, unsuitable microclimate and/or the role of regular fire which maintains the grass domination (the fire frequency being intermediate between that of Baluran and the other savannas studied). The dominant grass here (*Imperata cylindrica*) is known for its fire tolerance and is considered a fire-climax species across Southeast Asia (MacDonald 2004). Although the Rinjani savanna studied is not currently shifting to a forest state, of the woody species found in my plots at Rinjani, almost all of them are forest pioneer/edge specialists (Appendix 1), many currently occurring as seedlings or saplings. This indicates invasion from nearby gallery forests in nearby drainage lines and the potential for conversion to forests in the absence of future fires over a long period. However the clumps of trees and small forest patches which establish in crevices and drainage lines are likely promoted by the extra moisture and conducive microclimate conditions which encourage dense woody vegetation (Murphy and Bowman 2012). In such landscapes, forests are often constrained to geomorphology/topographic conditions that protected it from fire (Russell-Smith *et al.* 2012).

Tropical savannas generally have high alpha (local-scale) plant species diversity, particularly when compared with temperate grasslands and dry tropical woodlands (Solbrig *et al.* 1996). High diversity in savannas for instance occurs in the Brazilian cerrado (Furley 1999). However, savanna diversity can be reduced due to several reasons, such as invasion of exotic species, fire exclusion, herbivore introduction or exclusion, or physical removal of trees and shrubs (Furley 1999; Solbrig *et al.* 1996). In my study, average species diversity is in the low to moderate range for savanna (Magurran 1988; 2004). Moderate species diversity is also found in other Asian savanna, such as in India (Pandey and Singh 1991). In Indonesia, such low to moderate diversity is also known to occur in the wet-climate savanna at Pangandaran, West Java (Rosleine and Suzuki 2013). This relative paucity of species in these savannas is perhaps due to extensive invasion of exotic species and fire exclusion. At Pangandaran, savanna abandonment without recurring fire has resulted in over-dominance of *Decaspermum* spp. which then caused the drop in species diversity (Rosleine and Suzuki 2013). Low species evenness may also be the result of the dominance of certain exotic species (both woody and non woody) in the other savannas studied (i.e. *Acacia nilotica* in Baluran savanna, *Lantana camara* in Bali Barat, and *Chromolaena odorata* in Alas Purwo). In their study in Baluran, Caesariantika *et al*. (2011) found that *Acacia nilotica* invasion had significant effect on species diversity in Baluran savanna, with values of (Shannon-Weiner) species diversity in the range of 0.46 to 1.34. In my study, I found broadly similar mean Shannon-Weiner species diversity of 0.74 at Baluran. In Baluran, *Acacia nilotica*, a woody IAS, is fire tolerant when mature and unpalatable due to considerable thorns (Djufri 2004; FAO 2014). This poses an increased threat of expansion of *Acacia nilotica* stands in Baluran savanna even with regular fire and high grazing pressure. Evidence of recent expansion of the *Acacia nilotica* stand in Bekol savanna, Baluran is outlined in Sutomo *et al*. (2016). Fire exclusion at Bali Barat and Alas Purwo may also play a role in the relatively low species diversity of these savannas. Although I sampled my savannas in the dry season, many annual and other short-lived species were still able to be identified as remnants of the flowers and/or leaves were still present on at least some plants. However I accept that I am likely to have under-estimated plant diversity, especially of annual forbs and geophytes, compared to if I have sampled late in the wet season.

In conclusion, this study is the first to thoroughly describe savanna plant communities in the wetter parts of the Indonesian archipelago. I have shown there are distinct elevation gradients (along with related climatic factors such as temperature and precipitation), as well as fire regime gradients, linked to tropical savanna community composition across Java, Bali and Lombok Islands which are characterized by different set of species, and where IAS are becoming significant components of the ecosystems. Therefore IAS management should be considered by these national parks authorities as it may lead to greater native species diversity. Lack of prescribed fire and a range of invasive species threaten to convert savanna at Bali Barat and Alas Purwo into secondary forests or shrubland, whereas the presence of forest pioneer/edge species within the savanna at Rinjani suggests successional change from grassland to forest may occur in the absence of future fires (although the role of soil, topography and microclimate in maintaining grass dominance needs also to be explored). More study of soil as a factor controlling savanna distribution and their species is also needed. Compared to the others studied, the savanna in Baluran National Park has characteristics of being relatively old and persistent (Ratnam *et al.* 2016), rather than one being created and maintained via human conversion of forests. It is recommended that further studies be conducted, including isotopic characterisation of soils and organic matter and manipulative experiments involving use of fire and grazing, to more firmly establish the dynamics and age of the savannas studied.

Chapter 3: Spatial and Temporal Patterns of Fires in Tropical Savannas of Indonesia

ABSTRACT

This chapter examines spatial and temporal patterns of fires in tropical savanna located across the Indonesian Archipelago. It is hypothesized that fire is more prevalent in savanna landscapes compared to other types of landscape, and that fire incidents are more common during the dry season and drought periods, as well as being more prominent in the drier parts of Indonesia Archipelago compared to the wetter regions. Remote sensing was used for these purposes, particularly the MODIS burned area product over a 14 years period (2000 to 2013 inclusive), used in conjunction with climatic and land use data for the Indonesia region. Image display and data processing and analysis were conducted using ARC-GIS. Results showed that across Indonesia, fires (burnt area and number of fires) were detected the most in year 2002 and were least in 2010. The extent of burning was found to be correlated with the annual Sothern Oscillation Index (Spearman's Rho = -0.604, P value = 0.0221) with most burning occurring during times of sustained negative SOI values, which generally means drier conditions across SE Asia. Across Indonesia, most of the detected fires occurred in the middle of the dry season (August-September). Fires were also more common in the drier parts of the Indonesian Archipelago, with most burning occurring in East Nusa Tenggara (NTT) Province, particularly Sumba, Sumbawa and Timor Islands. On Sumba Island, where the dry season is particularly severe and extends for at least four months, savannas/open vegetation is the most common land cover type, of which 99% was burnt by area over the study period making this the most burnt of major islands. Overall in Indonesia, there is much less savanna compared to various types of forest, hence it is expected that a greater percentage of this will be burned. Indeed, result showed that approximately 2% of savanna/open vegetation had burned whereas only 0.8% of forest has burned by area over the period 2000-2014. Much of the savanna and other open vegetation in the region are also active grazing lands so better understanding of the interactions between fire and grazing, and their impacts on productivity and biodiversity, should be a research priority. El Nino forecasting could be important for fire planning in the region, given far more fires and land area is burnt during these periods.

Keywords: Fires, savanna, remote sensing, Indonesia, MODIS, ENSO, Baluran National Park, Sumba, Tambora.

3.1. INTRODUCTION

Although savannas only occupy 20% of the global land surface, they account for 30% of terrestrial net primary production and most (~50-70%) of the annual global burned area (Lehmann *et al.* 2014a). Chang and Song (2009) estimated that around 50% of savanna burns annually, whereas Giglio *et al.* (2013) reported that 71% of annual global burn area is savanna. Fires in savanna are either of anthropogenic origin (either deliberately lit for land management, arson or cultural purposes, or accidental) or natural (usually started by lightning) (Tacconi and Ruchiat 2006). For savanna or grasslands located on the slopes of volcanic mountains, fire can also be naturally lit by volcanic activity such as pyroclastic flows (*nuees ardentes*) or other events where fallen rocks spark fire (van Steenis 1972). Areas of recent lava flows or ash deposition are often first colonised by fast-growing grass (such as *Imperata cylindrica* in Asia) which can act fuel for fires, which may then develop into a tight fire-grass cycle where regular fires promote grasses and vice versa (Sutomo *et al.* 2015; van Steenis 1972). In general, regular fires in the tropics are widely believed to be important in creating and maintaining savanna (more open woody vegetation with grassy understory), especially in regions which, based on climate alone, should be forest (Bond and Keeley 2005).

Remote sensing (RS) and Geographic information systems (GIS) have been commonly used to map and analyse the fire history and spatial patterns of fires within landscapes and regions. Avitabile *et al.* (2013) used Landsat imagery to detect and map the fire scars of the Murray Mallee region of south-eastern Australia from 1972 to 2007, which allowed them to characterise spatial and temporal patterns of fires. A series of fire scars was mapped using all available Landsat TM images between 1989 and 2001 for an area of 63000 km^2 in north-east Namibia to investigate the relationship between fire frequency, rainfall, and land cover (Verlinden and Laamanen 2006). Other researchers have utilized remote sensing data derived from MODIS satellite sensors to map and analyse fire history and patterns of fire scars of areas (Chongo *et al.* 2007; Csiszar *et al.* 2005; Devineau *et al.* 2010; Tewes *et al.* 2015). Two fire products are derived from MODIS data: MODIS active fire product which is based on detection of current fires, and MODIS burned area product which detects recently burned patches of vegetation or fire scars. Devineau *et al*. (2010) showed that using a combination of both can be helpful to understand the link between fire and landscape type in Sudanese savannas. Spatial and temporal analysis of savanna fires in Africa has also been done by Cahoon (1992) where it was observed that almost all of savanna fires resulted from human activities and most fires were left to burn uncontrolled so that there was no strong seasonal cycle in the fire frequency. This research demonstrates the value of combining region-wide vegetation mapping with fire history mapping. Systematic and consistent mapping of fires in fire-prone regions is an essential foundation for improved fire management and more effective landscape planning for conservation (Avitabile *et al.* 2013; Beatty *et al.* 2015).

There are two broad approaches to detect fires using remote sensing: 1) detect active fires from thermal or light signatures (e.g. infra-red reflectance), which are often called fire ‘hot spots’; 2) detect fire scars on recently burnt ground by detecting rapid change in vegetation cover from successive images combined with some other evidence of fire (e.g. vegetation type, seasonality or presence of charcoal), which are known as ‘burned areas’ (Giglio 2015; Giglio *et al.* 2006). There are various data sources to use and each has its own limitations. Burned areas mapping has used a variety of sensors such as Advanced Very High Resolution Radiometer (AVHRR) deployed on NOAA satellites, Landsat and, in more recent times, MODIS and Himawari-8 (Bowman *et al.* 2003; Buthelezi *et al.* 2016; Franca and Setzer 2001; Giglio 2015). The choice of these sensors basically depends on the scale and resolution at which fires need to be mapped (Buthelezi *et al.* 2016). For example, for regional scales, AVHRR which is characterised by a spatial resolution of 1000 m and a daily temporal resolution, has been demonstrated to be proficient (Franca and Setzer 2001; Mitri and Gitas 2013). Nevertheless, the low thermal sensitivity and low spatial resolution of the sensor hinders the accurate detection of fires that are less than its pixel size, thus underestimating the burnt area (Marrinan *et al.* 2005; Mitri and Gitas 2013). MODIS (Moderate-resolution imaging spectroradiometer) overcame this hindrance associated with AVHRR by providing a trade-off between fire mapping over time and space, offering greater resolution but being not as frequent (Buthelezi *et al.* 2016; Martín *et al.* 2002). Additionally, the temporal resolution of MODIS makes it a better source of burned area data when compared to other sensors such as Landsat with a 16-day temporal resolution (although Landsat has finer resolution compared with MODIS) (Buthelezi *et al.* 2016; Tsela *et al.* 2014). The Terra MODIS satellite passes from north to south across the equator in the morning, while Aqua MODIS passes south to north over the equator in the afternoon. Terra and Aqua MODIS are viewing the entire Earth's surface every 1 to 2 days, acquiring data in 36 spectral bands or groups of wavelengths (Maccherone and Frazier 2014). MODIS permits a long time series of burnt area mapping, given of its high temporal resolution (i.e. running since 1999) compared to many other available sensors (Buthelezi *et al.* 2016).

Bowman *et al.* (2003) evaluated four methods to map fire scars apparent on Landsat-TM imagery of tropical savanna in northern Australia. They concluded that Landsat-TM imagery cannot be used to reliably determine the spatial extent and timing of fires in environments with rapid post fire recovery, such as tropical savannas, thereby limiting the utility of this data source for fine-scale ecological studies. Multiyear global burned area products have been released for public use, namely Collection 5 of the MODIS burned area product (MCD45A1) (Roy *et al.* 2008). This MODIS burn area product (MCD45A1) according to Chang and Song (2009) is more comparable to reference data than the L3JRC burned area which is based on direct detection by the SPOT VEGETATION sensor. MODIS data has a fairly coarse resolution (from 250m to 1km) but has the advantage of quality, coverage, frequency and long time sequence. MODIS active fires detect all thermal abnormalities which can include fires and flares. Each “fire pixel” or fire detection in MODIS active fire covers one square kilometre, indicating that in the corresponding one square kilometre area on the ground, one or more fire is burning. In addition, MODIS burned area product includes a charcoal detection as well as change of vegetation cover in its algorithm. The MODIS burned area product has been shown to be highly effective and accurate in mapping areas burned by larger vegetation fires, although it can underestimate the extent and number of ground and sub-surface fires (e.g. peat fires) in dense forest (Roy *et al.* 2008).

For more than two decades, there has been rising concern over the environmental impacts of biomass burning in the wet forests of Sumatra and Kalimantan in Indonesia (Russel-Smith *et al.* 2000). This has translated into a substantial research endeavour in these regions (Aiken 2004; Brown 1998; Byron and Shepherd 1998; Clearly *et al.* 2006; Lavigne and Gunnell 2006; Ruchiat 2001; Simbolon *et al.* 2003b). In contrast, much less consideration has been given to more frequent burning practices and linked land management problems across the savanna ecosystems of the Indonesian archipelago. Globally, there has been many studies using RS/GIS to detect and analyse fire in savanna ecosystems (Chacón-Moreno 2004; Hudak and Brockett 2004; Sano *et al.* 2010; Stroppiana *et al.* 2003), however this is not the case in Indonesia, which is surprising given widespread distribution of savannas in eastern Indonesia, such as the province of East Nusa Tenggara (Indonesian: Nusa Tenggara Timur; NTT). One of the few studies of fire using RS/GIS in Indonesian savanna was done by Fisher et al. (2006) in Flores and Sumba Islands, NTT, who applied fire mapping techniques using Landsat imagery to several locations in east Sumba and central Flores, but only for two years and in relatively small areas surrounding villages.

Here I explore the patterns of medium to large vegetation fires across the Indonesian archipelago using the MODIS burned area and active fire products in the years they were available (2000-2013). I hypothesized that fire is more common in savanna landscapes compared to other types of vegetation, and that fire incidents are more prominent following the dry season, particularly during or immediately following periods of extended drought in Indonesia. I was also interested in establishing if fires were more prevalent in areas of volcanic terrain. Remote sensing could provide insight into the fire patterns and its importance to land management especially in savanna landscapes, particularly in eastern Indonesia where savannas are widespread (Tacconi and Ruchiat 2006).

3.2. MATERIALS AND METHODS

Indonesian Terrain Feature Map (Peta Rupa Bumi Indonesia/RBI) which consist of topography, land use and administrative map sheets (1:50,000) was acquired from Indonesian Geospatial Agency/*Badan Koordinasi Survey dan Pemetaan Nasional Indonesia* (BAKOSURTANAL/BIG) as both hard copies and digital coverages. Table 1 provide details of the RBI maps. These formed the base layer for spatial alignment of all other spatial data. This base layer used a Plate-Carree projection with WGS84 datum, with all other coverages align to this map.

Table 1. Characteristics of the Indonesian Terrain Feature Map (Peta Rupa Bumi Indonesia) used as basic land use/type map

Type of information	Remarks
Issued by	Indonesian Geospatial Agency (Badan Infomasi Geospasial/BIG)
Method	Aerial photography, interpreted by team of BIG experts
Resolution	30 x 30 m
Validation	80% accuracy, tested using confusion matrix
Year made	2013
Update	Every 10 years
Scale	1 : 50,000

I also made use of a map of climatic regions of Indonesia derived from Aldrian and Susanto (2003) which defined three climate regions: Region A, the southern monsoonal region (South Indonesia, from South Sumatera to Timor Island, South Kalimantan, Sulawesi Island and some parts of West Papua); Region B, the semi-monsoonal region (Indonesia Southwest, northern part of Sumatera, Northeast of Kalimantan); and Region C, the anti-monsoonal region (Moluccas and some parts of Sulawesi). Region A corresponds to areas with maximum rainfall falling in December/January/February and minimum in July/August/September. Region B has two peaks of wet season annually: October/November/December and March/April/May, whereas region C only has one peak which is in May/June/July (Aldrian and Susanto 2003; Hanifah 2014). Southern Oscillation Index (SOI) is defined as the normalized pressure difference between Tahiti and Darwin. Here I made use of SOI data from the Climatic Research Unit (https://crudata.uea.ac.uk/cru/data/soi/), which is based on the method given by Ropelewski and Jones (1987).

Moderate Resolution Imaging Spectroradiometer (MODIS) products were used to detect fire events (MODIS active fire product: MOD14) and broad fire coverage

(MODIS Collection 5.1 Global Burned Area Product: MCD45) across the Indonesia archipelago for the period of 2000 - 2013. However it is noted that south-eastern tip of Sumba Island is not included in Tile 19 (which covers most of SE Asia) as as well as some other very small islands to the south of Sumba. The MODIS Burned Area Product MCD45 has 500 m resolution whereas the MODIS active fire MOD14 has 1 km resolution. This data was obtained in a form of Arc GIS shape files from the University of Maryland (http://modis-fire.umd.edu/index.php). Monthly collated data was obtained and then merged into annual coverages. These were then merged into overall coverages for years 2000 – 2013. For accuracy of burn date, I agreed with Boschetti *et al*. (2013) and Roy *et al.* (2008) that in terms of the burn date in the MODIS product, 50% reflect fires occurring within the previous day and 75% reflect fires within the previous four days. For much of the study area, the median error is 1 day.

Active fires are located on the basis of the so-called thermal anomalies produced by them. The algorithms compare the temperature of a potential fire with the temperature of the land cover around it; if the difference in temperature is above a given threshold, the potential fire is confirmed as an active fire or "hotspot" (EFFIS 2015). In this study, the term “hotspot” refers to detection of active fires (i.e. one or more fires that detected in 1 km grids). I established spatial distribution of fires in Indonesia for period of 2000 – 2013 by overlaying our base maps with the MODIS data using ARC MAP 10.1. I then quantified the data extracted from the created maps namely: number of fire hotspots per month and per year and per province for the specified period, and the estimated size of burned areas for different types of land use per year for the specified period. Fire “hotspots” from MODIS data were then tabulated based on year, month, area/size per land use, and province using the table feature in Arc MAP and imported to Microsoft Excel for further analysis. In terms of land use, I used open/very sparse vegetation to delineate areas of savanna, but with caution as this land class may also include high-altitude grassland and abandoned land. Other land use categories were similar to those used by Stibig *et al.* (2003). Greater attention was given to Baluran National Park in East Java, Tambora National Park in Sumbawa West Nusa Tenggara, and Sumba Island East Nusa Tenggara Indonesia due to known prominent extent of savanna fires. We also examine correlations between drought time/period (annual average SOI index) with the annual occurrence of hotspots and area burned. In addition, I overlaid burned areas detected with a map of volcanoes (extinct and active) that stretched from West Java to Tambora East Nusa Tenggara to look for evidence that fires were associated volcanic terrain. I matched the year when burned with the known eruption dates of these volcanoes. Ground-truthing was previously conducted for several sites at Baluran National Park in East Java as well as Rinjani National Park in Lombok Island West Nusa Tenggara in 2014 and the Kupang Province on Timor Island East Nusa Tenggara in 2015. Ground-truthing was done by comparing previous collected field data on fires from these sites with MODIS derived burn area coverage in GIS. In Baluran, for instance, I have created a fire map of Bekol Savanna using Landsat (Chapter 5 and 6) and it is seemed that MODIS missed many small fires at Bekol due to park managers only burning one small block of savanna at a time.

3.3. RESULTS

The number of hotspots was high in 2002 and 2006 with the 2002 (4,554 detections) being the highest (Table 2). Hotspot detection in this year peaked in October (1,752 detections). The number of hotspots was lowest in 2010 (350), with 2000 and 2013 also being relatively low. In 2010, hotspots detection was highest in the months

June to October, peaking in August (272). In terms of hotspot detections per month (during the 2000-2013 period), August was the highest month with 7,812 detections and was just slightly higher than September (7,237). Very few active fires were detected in the months January to March with February being the lowest point with only nine fire hotspots detected (Table 2).

Table 2. Temporal distribution of hotspots (number of 1km^2 grid cells with at least one active fire detected in the month) in Indonesia for each month over a fourteen year period (2000 to 2013).

Year	**Month**												
	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sept	Oct	Nov	Dec	∑
2000	0	0	0	0	2	91	153	89	323	10	0	0	668
2001	0	1	0	3	14	0	130	402	358	30	0	0	938
2002	0	0	0	7	48	326	287	949	1166	1752	17	2	4,554
2003	0	0	6	10	69	252	368	417	280	143	9	1	1,555
2004	0	0	0	7	27	218	384	1052	517	539	30	34	2,808
2005	22	3	7	4	102	178	183	349	297	34	1	0	1,180
2006	0	0	0	1	13	81	663	1125	1162	844	230	3	4,122
2007	0	2	2	2	11	100	182	439	1266	183	0	0	2,187
2008	1	0	0	3	89	171	484	463	360	72	0	0	1,643
2009	1	0	1	12	9	226	428	571	419	210	21	1	1,899
2010	0	3	2	1	1	12	51	272	8	0	0	0	350
2011	2	0	0	0	59	296	314	443	232	14	1	1	1,362
2012	0	0	0	2	21	99	476	996	630	51	0	0	2,275
2013	0	0	1	2	3	2	58	245	219	38	0	0	568
SUM	26	9	19	44	468	2,052	4,141	7,812	7,237	3,920	309	42	26,109

In line with the number of fire hotspots detected per year, burned area per year follows a similar pattern. Burned area was highest in 2002 with total area detected as being burned 240,553 ha in total detected as being burned. The lowest detection was in 2010 with 14,550 ha burned (Table 3).

Table 3. Burned area per year in Indonesia from 2000 to 2013 as detected using MODIS burned area product.

Year	**Size (Ha)**
2000	24,337.9
2001	38,152.3
2002	240,552.8
2003	61,322.6
2004	125,076.1
2005	49,631.3
2006	179,590.5
2007	94,871.8
2008	65,339.9
2009	84,416.2
2010	14,549.6
2011	54,510.9
2012	95,091.9
2013	8,0420.7

A map of the main land use type is shown in Figure 1. The percentage land use burned relative to the availability of the land use type is shown in Table 4. There is much less savanna than forest (forest covers as much as 55% of the total land area, whereas savanna cover is about half of that of forest), so it is not surprising that a (slightly) lower amount of this land type has burned. However I found that almost 2% of the available savanna/sparseland area had burned, but only 0.8% of the available forest had burned by area (Table 4).

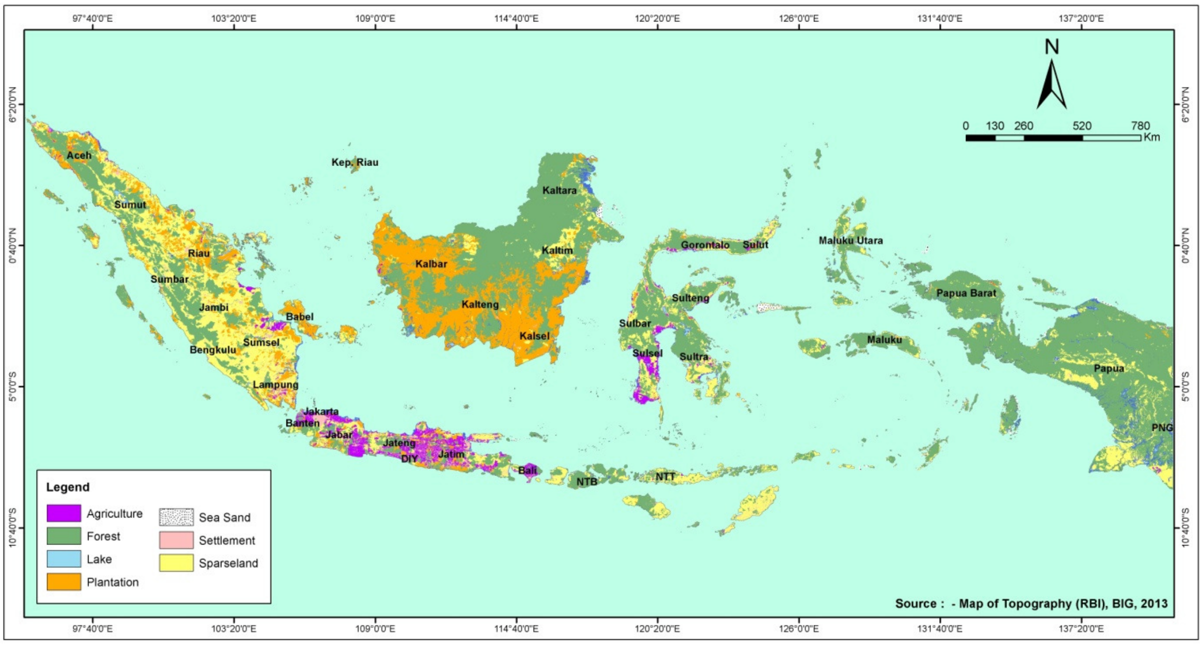


Figure 1. Map showing main land use type across the archipelago in Indonesia

Table 4. Burned area by land use (from year 2000 to 2013) in Indonesia. Last column indicates proportion of the land use type that is burnt (percentage of land use burned divided by percentage of land use). Savanna/sparseland correlation with SOI = -0.7 ($p < 0.001$) whereas forest correlation with SOI = -0.8 ($p < 0.001$). Note, SOI is Southern Oscillation Index - strongly negative values tend to mean drought over SE Asia.

Land use	**Size (ha)**	**% of total land use**	**Total land use burned (ha)**	**% of total land use burnt**	**Proportion of the land use type that is burnt**
Agriculture	62356.7	3.13	7202.96	5.96	1.9
Forest	1101700.7	55.44	53653.51	44.42	0.8
Freshwater	19991.4	1.01	1603.58	1.32	1.3
Plantations	299059.4	15.05	4991.76	4.13	0.27
Coastal	17870.9	0.89	138.68	0.11	0.12
Settlement	45134.3	2.27	1406.90	1.16	0.51
Savanna/Sparseland	440772.6	22.18	51789.08	42.87	1.93

Each main island of the archipelago has different total number of fire hotspot detection (Table 5). The highest was detected in East Nusa Tenggara (NTT) Province. East Nusa Tenggara consists of several islands, with the main ones being Flores, Komodo and Sumba. In this province fire hotspots detected reached 14,918, with Sumba and Timor Islands being the most prominent ones in terms of fire occurence. Bali Island was the lowest with only 47 hotspots detected.

Table 5. Summary of fire hotspots distribution in Indonesia (2000 – 2013) by major island groups

Island/ group of Island/ Province	**Land area (Km^2)**	**∑ Hotspots**
Kalimantan	537010	584
Papua	469698	1,780
Jawa	134806	4,618
Sulawesi	187376	1,316
Sumatera	512652	240
NTB	19694	2,665
NTT	46811	14,918
Bali	5586	47

Figure 2 displays the climatic regions of Indonesia based primarily on the seasonality and timing of rain, with fire hot spots overlaid. Most of the hotspots were detected in the A climatic region (Figure 2) which corresponds to the parts of Indonesia with pronounced dry season where maximum rainfall occurs in December to February, and minimum from July to September. The map also confirmed the results presented in Table 1 where, in terms of active fire detection patterns observed in the period of 2000 to 2013, August was the month with most fires (7,812 hotspots), whereas February was the month with the fewest fires with only nine hotspots detected.

Figure 3 showed temporal pattern of total number of hotspots per year together with annual SOI for the period of 2000 to 2013 in Indonesia. The graph showed that when the total number of fire hotspots detection was high, the SOI value tended to be low. Total number of hotspots per year was negatively correlated to mean annual Southern Oscillation Index (SOI; $r = -0.604$, $p<0.05$). Sustained negative SOI values are indicative of an El Nino climatic phase, which generally means drier conditions across SE Asia.

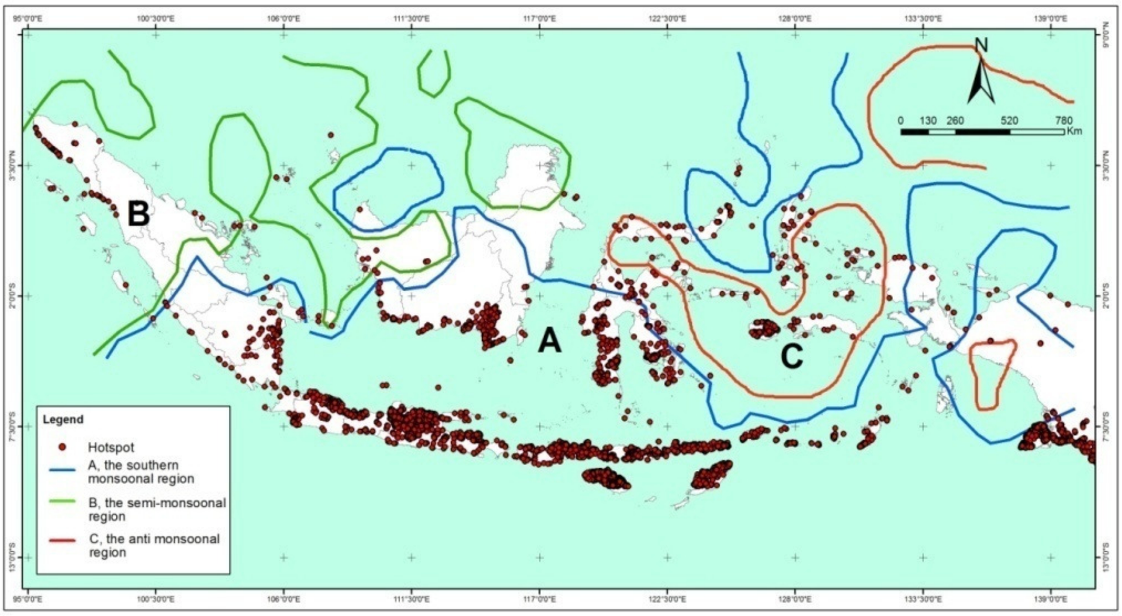


Figure 2. Map showing overlay of fire hotspots during 2000-2013 period, with three climate regions in Indonesia. Red dots are 1 km grid square with at least one active fire detected over the study period.

Figure 3. Correlations between drought time/period (SOI index and average precipitation) with the occurrence of fire hotspots (i.e. number of 1 km^2 grids with fire recorded during the study period) for each year 2000-2013 in Indonesia. Correlation coefficient = -0.604, P value = 0.0221

Sumba Island in East Nusa Tenggara is most prominent in terms of fire hotspots and burned area detected. Sumba Island is located in the region A (southern monsoonal region) of Indonesian climate classification region and experiences a generally drier and more strongly seasonal climate than elsewhere in Indonesia, with most vegetation being savanna (Aldrian and Susanto 2003; Hanifah 2014; Monk *et al.* 2000). Figure 3 shows the spatial pattern of hotspot distribution on Sumba Island from 2000 to 2013 except for south-eastern tip of Sumba Island which is not included in Tile 19 of the MODIS data. Hotspots were mostly detected along the northern coastline of Sumba Island and in the district of East Sumba wherethere is much savanna country/open vegetation relative to forests on the island (Figure 4). August is the month where total number of fire hotspots detection and burnt areas were at their highest level.

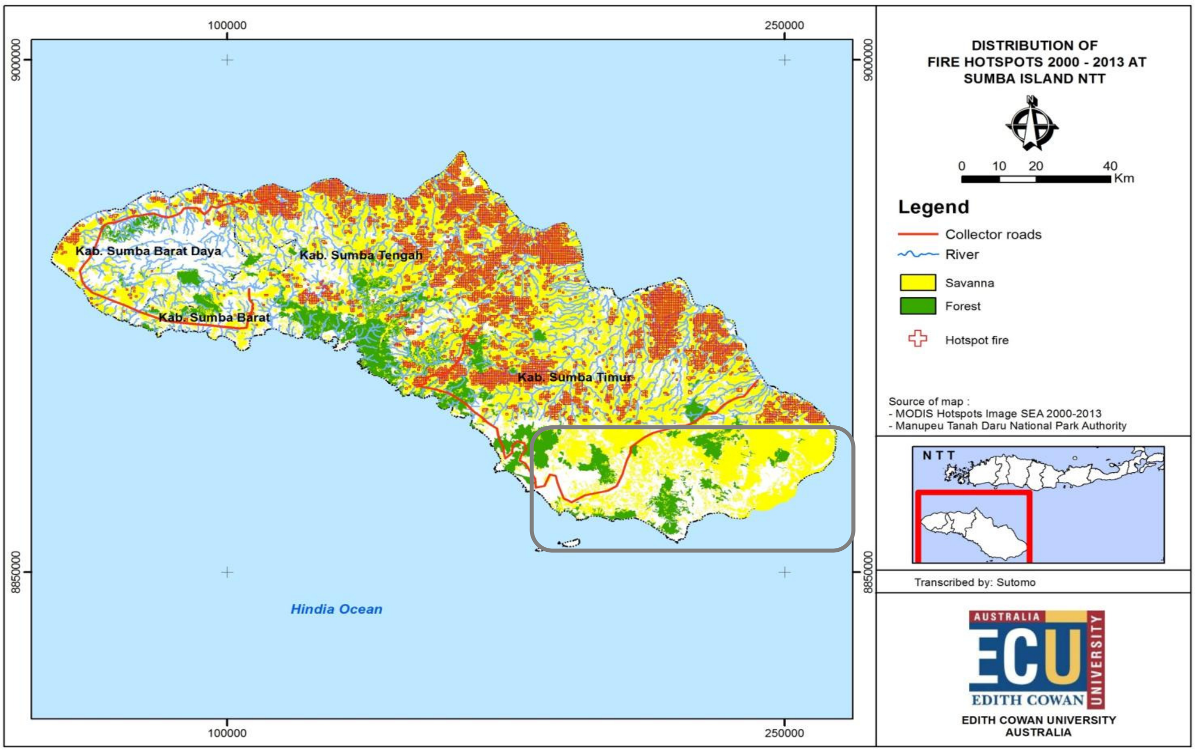
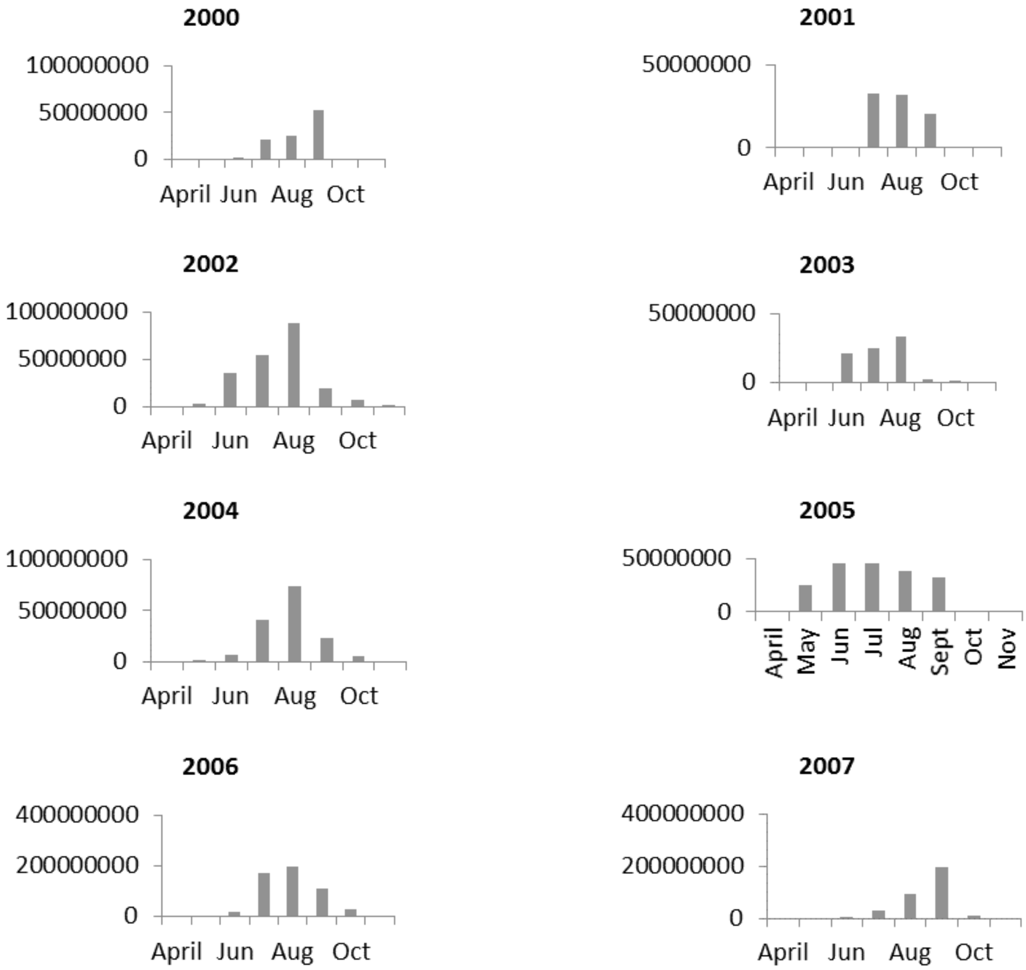


Figure 4. Fire hotspot distribution on Sumba Island, East Nusa Tenggara (NTT) from 2000 to 2013 (south eastern Sumba is excluded as Tile 19 of MODIS data did not cover this section – grey bounded rectangle). In this map, the term of hotspots are shown by the orange shading, which is a combined burn areas showing areas burned at least once during 2000-13 using MODIS burn area product.

On Sumba Island, savanna burning comprises about 99% of the total area burned, whereas forest was just 1 % of the total area burned. Savanna was burned the most in 2006 (51,855 ha) and was at the lowest point in 2001 (8,594 ha) (Table 6). Whereas forest burned areas was at the highest level in 2008 (221 ha) and lowest in 2000 (14 ha). On average, for the period 2000 to 2013, fires occur between months of May to October, with 2002, 2008, 2009 and 2013 being exceptional in that fires also occured in April and November. The years 2001, 2008 and 2009 were the years where Sumba Island had 7 months of savanna fire (Figure 5).



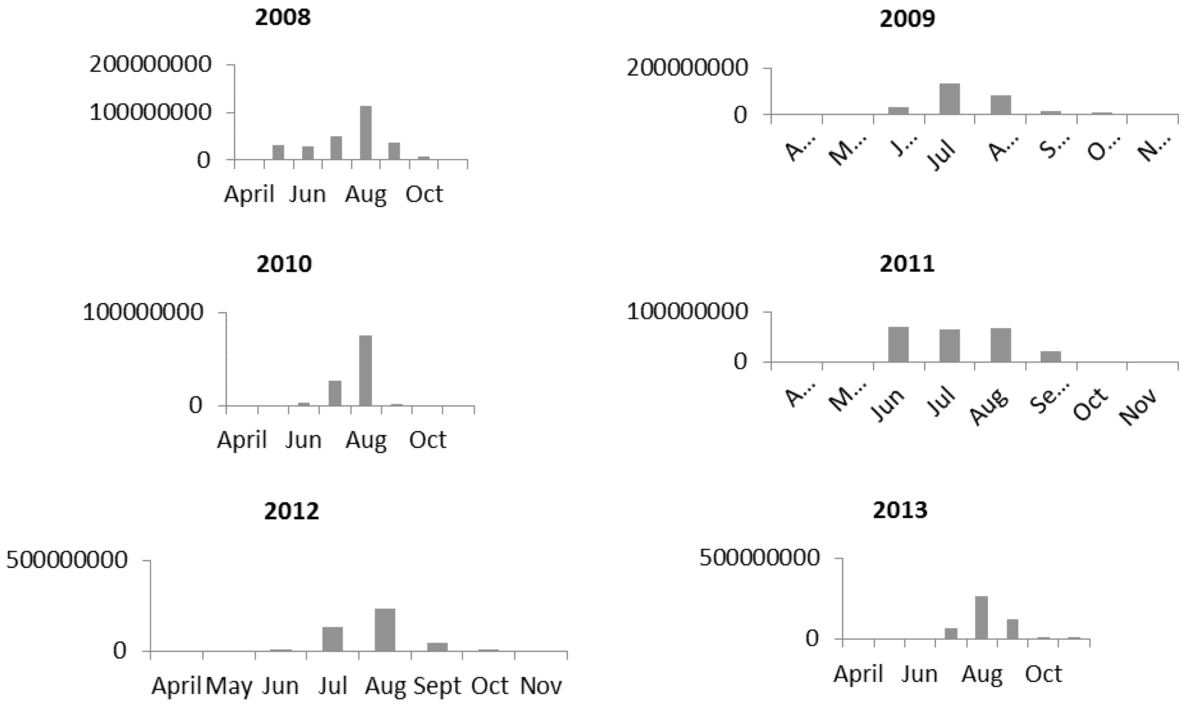


Figure 5. Yearly pattern of savanna burned (ha) per month in the period of 2000 to 2013 on Sumba Island, East Nusa Tenggara Indonesia (NTT).

Table 6. Savanna and forest areas burned in the period of 2000 to 2013 on Sumba Island, East Nusa Tenggara Indonesia (south eastern tip of Sumba is excluded).

Year	**Areas burnt (Ha)**	
	Savanna	**Forest**
2000	10003.67	14.02
2001	8593	121.22
2002	20756	67.41
2003	9873	25.40
2004	15114	107.81
2005	18689	47.65
2006	51855	170.26
2007	33381	109.98
2008	26508	221.10
2009	28413	137.58
2010	10645	48,67
2011	22439	68.95
2012	42662	164.80
2013	45530	83.85
Total	344,470	1,388.7

There were some spatial association between areas that burned often and their occurence on or near volcanic mountains along the chain of islands from Java, Bali, and Lombok to Sumbawa Islands (Figure 6). Most of these areas were located near volcanos and they burnt very frequently. On Java Island, some of burned areas were observed near Merapi, Raung, and Baluran mountains (Central and East Java/area 2 and 3). To the east, burned areas were also observed near Mount Agung in Bali, Mount Rinjani in Lombok (area 4) and prominent burned areas were spotted near Mount Tambora in Sumbawa (area 5) (Figure 7). Some but not all of these burned-areas years are matched with the year when volcanic eruption occurred (Table 7). Recent hotspots detected near Agung and Tambora were most likely forest and or savanna fires (Figure 6) as the last known eruptions of these two volcanoes was over 50 years ago (Agung) and more than a century ago for Tambora (103 years) (Table 6).

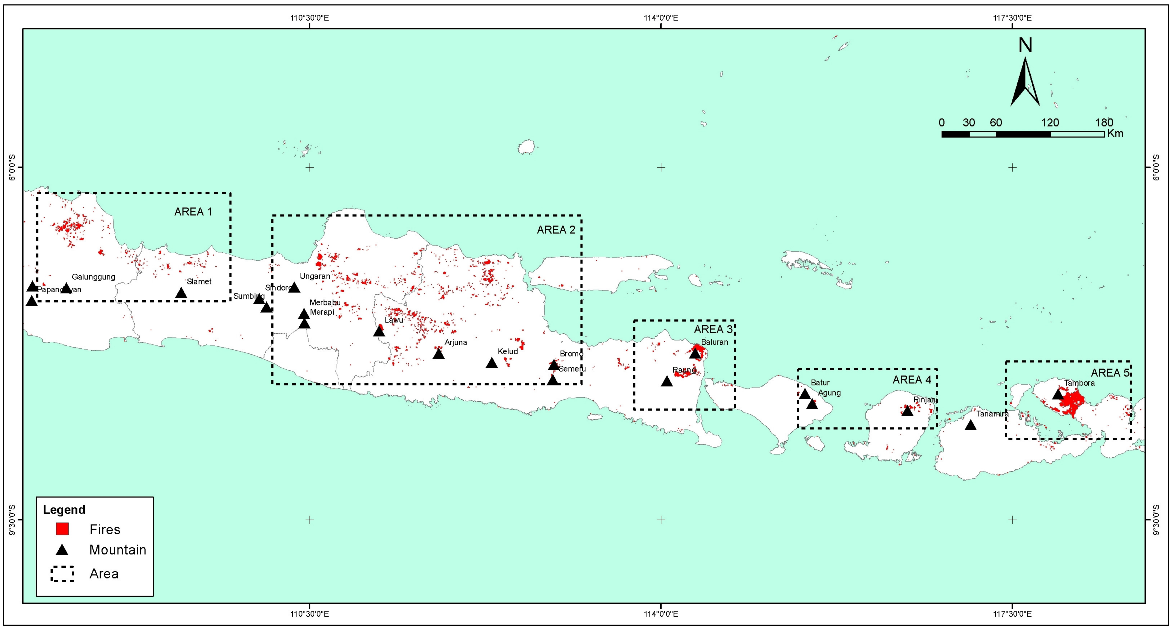


Figure 6. Distribution of volcanic mountains and burned area along Java, Bali, Lombok and Sumbawa Islands Indonesia

Table 7. Years when fires detected by MODIS near volcanoes and their last known eruptions.

Region/area	Known large fires (MODIS)	Vocanoes presence nearby	Recent known eruption	Reference
Area 1 West Java				
	2012	Papandayan	2002	Sutomo and Fardilla (2012)
	2002 & 2006	Tangkuban Parahu	2002, 2005, & 2013	Ari (2013)
Area 2 Central Java				
	2002 & 2006	Merapi, Merbabu	2002, 2006 & 2010	(MVO 2006; Sutomo 2013)
	2002 & 2011	Lawu	1885	(GVP 2013)
	2002 & 2006	Arjuna	Unknown	(VSI 2014)
	2002	Bromo	2000, 2004, & 2010	(VSI 2014)
	2011	Semeru	2002, 2004, 2005, 2007 & 2008	(VSI 2014)
Area3 East Java				
	2001, 2002, 2003, 2004, 2005, 2006, 2007, 2008, 2009, 2011, 2012 & 2013	Baluran	Unknown	(BTNB 2010; GVP 2013)
	2001, 2002, 2004, 2006, 2007, 2008, 2010, 2011, 2012 & 2013	Raung	2000, 2002, 2004, 2005, 2007, 2012, & 2015	(VSI 2014)
Area 4 Bali-Lombok				
	2004, 2006, 2007, 2008, 2012, 2013	Agung	1963	(ESDM 2014a; Sutomo and Fardila 2012)
	2002, 2005, 2006, 2007, 2008, 2012	Batur	2000	(GeoMagz 2012)
	2004, 2006, 2007, 2008, 2009, 2011, 2012	Rinjani	2004 and 2009	(ESDM 2014b; Khafid 2009)
Area 5 Sumbawa				
	2003, 2004, 2005, 2006, 2007, 2008, 2009, 2010, 2011, 2012, 2013	Tambora	1815, 1913	(ESDM 2014c)

On Sumbawa Island, most savannas are situated on the slope of Mount Tambora volcano and bordered with adjacent forest (Figure 7). Hotspots were detected in these areas in all year between 2003 to 2013 (Table 8). On Sumbawa Island, burned savanna made up 32.8% of the total area burned, whereas forest was 67.2% of the total area burned. Savanna was burned the most in 2007 (6,247.5 ha) and was at the lowest point in 2005 (164.6 ha) and there was no hotspot detected in savanna in 2010 (Table 8). Whereas forest burned areas was at the highest level in 2006 (12,049.9 ha) and lowest in 2010 (235.2 ha).However, there are far more forests than savanna and that at least 90% of savanna was burnt during the study period.

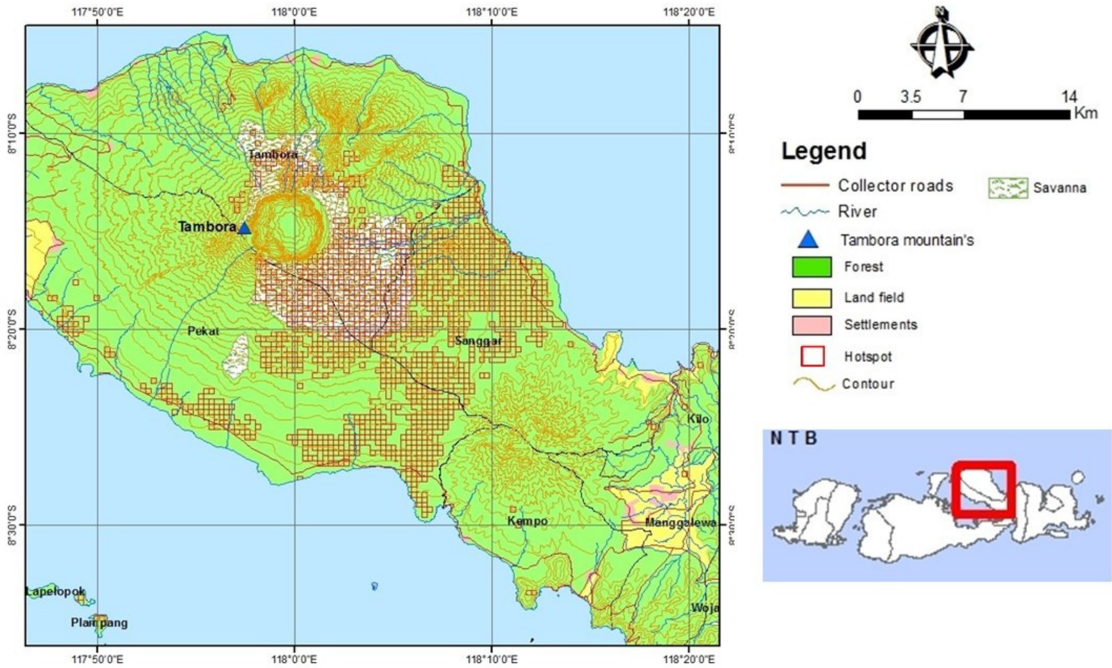


Figure 7. Combined burned areas showing areas burned at least once during 2000-13 using MODIS burn area productin the area around the Mount Tambora Volcano on Sumbawa Island, West Nusa Tenggara (NTB).

Table 8. Savanna and forest areas burned near Mount Tambora Volcano on Sumbawa Island, West Nusa Tenggara (NTB).

Year	Areas burned (Ha)	
	Savanna	Forest
2003	431.4	4,771.8
2004	3,949.4	9,640.3
2005	164.6	2,389.1
2006	4,790.1	12,049.9
2007	6,247.5	5,017.6
2008	2,534.8	4,057.4
2009	1,910.8	5,199.1
2010	-	235.2
2011	314	2,835.9
2012	2,700.1	7,693.1
2013	4,893.7	3,389.7

3.4. DISCUSSION

In this study I examined the patterns of medium to large vegetation fires over a 14 year period across the Indonesian archipelago using MODIS burned area and active fire products, and whether these fires were correlated with El-Nino and associated drought periods indicated by the annual Sothern Oscillation Index (SOI). Firstly, using MODIS products, the result shows that, across the 14 year study period, a total of 26,109 active fire hotspots (1 km grid with at least one fire occurred) were detected and as much as 1,207,866 ha were burned; most of the detected burned area were forests (44% of the total burnt area) and savanna (42%) compared to other land use types. However there is much more forest than savanna in Indonesia, with a far greater proportion of available savanna burning (~2% burned) compared to forest (0.8%). Savanna in this land use category also includes other open vegetation, and with this in mind, I found that when I examined such areas of open vegetation with many fires, they all are known savanna (e.g. Baluran, Sumba and Mt. Tambora). Secondly, annual total number of active fire hotspots was negatively correlated to mean annual Southern Oscillation Index - sustained negative SOI values are indicative of an El Nino climatic phase, which generally means drier conditions across SE Asia. This support other studies showing more fires in Indonesia and Southeast Asia in El-Nino years (Byron and Shepherd 1998; Fuller and Murphy 2006; Mitani *et al.* 2009; Murphy 2006; Siegert and Hoffmann 2000; Simbolon *et al.* 2003b). However, these studies focus on fires in rainforest areas. Nonetheless, I have also demonstrated that fires in savanna also follow the same pattern. Most of the fires occurred in climatic region A (southern monsoonal region) of the Indonesian Climatic Classification where the peak of dry season is in July, August and September. Thirdly, East Nusa Tenggara islands (NTT) was the province/islands where most fires were detected. On Sumba Island East Nusa Tenggara (NTT), savanna comprises the largest land cover and accounted for more than 90% of the total burnt area, with August being the peak burning month, corresponding to the early-mid dry season. Lastly, there were associations between areas burned frequently and areas on or near volcanic mountains along the chain of islands spanning Java, Bali, Lombok and Sumbawa Islands; although some of these burned areas years are matched with the year when fires and or volcanic eruption occurred, most fires are not associated with such phenomena.

This study showed that Java Island is the second in the list after East Nusa Tenggara in terms of total number of hotspots detected during the period of 2000 to 2013, which is even higher compared to larger regions of Papua, Sumatra and Kalimantan. Java Island is interesting, as Lavigne and Gunnel (2006) point out, in that it represents what they termed as “inner” Indonesia, with only 7 % of the land area and 3% of the forest, but with 67% of the population. In their study of abrupt environmental and societal processes occurring on Java since the 1990s, using remote sensing on Javan volcanoes, they found that most of the fire extent detected was associated with natural (volcanic activity, ENSO) and human activities, with year 1994, 1995, 1997, 2001 and 2002 being the notable for years where land forest cover significantly decreased which was at least partly attributed to fires fires on the slopes of volcanoes (Lavigne and Gunnell 2006). Van Steenis (1972) noted the appearance of grasslands dominated by *Imperata cylindrica* (Cogon Grass) on mountain areas or in the former highland forests in Java (such as at Mount Panderman, Pujon and Mount Gilap in Iyang Highland in 1902 and 1940) following fires in forest and shrublands. I found frequent and relative large fires on the tops, slopes and surrounds of many of the volcanic mountains of Java, both active and dormant. Another cause of fires on Java Island is prescribed burning. Savanna in this wetter part of Indonesia such as Baluran National Park (Figure 7) and Alas Purwo National Park in East Java and Pangandaran Nature Reserve in West Java require regular burning to maintain open savanna vegetation which will transition to forest with fire exclusion (Rosleine and Suzuki 2013; Sutomo *et al.* 2015; Sutomo *et al.* 2016). Additionally, teak forest plantations owned by the State Forest Corporation (SFC/PERHUTANI) are also under prescribed and regular burning, such as those located in Central and East Java (Lavigne and Gunnell 2006; Sutomo *et al.* 2016).

Figure 7. Fire in *Bekol* Savanna Baluran National Park, East Java, dry season 2014

Montane forests such those in Java, Bali, Lombok and Sumbawa Islands are not resistant to fire. The forests are easily ignited under conditions of prolonged drought, such as when lightning strikes oil-rich species such as *Vaccinium* spp. On Mt Merapi in Central Java, pyroclastic flows or *nuées ardentes* are the primary cause of forest fire (Simon 1998; Whitten *et al.* 1996). *Nuees ardentes* caused extensive forest fires on the volcano slopes which may have change the successional trajectory to a more open grassland and or savanna (Sutomo and Fardila 2013; Sutomo *et al.* 2015). Recovery of the montane forest following fire is usually slow (Horn *et al.* 2001). Fire destroys the aboveground part of shrubs and some surviving species may be covered with ash, and with the characteristics of young volcanic soils this could slow or even arrest the rate of the secondary succession (Antos and Zobel 2005; Walker and del Moral 2009; Whitten *et al.* 1996). Severely burned areas on mountains in Java and Bali are usually characterized by the dominance of invasive species, such as *alang-alang* grass (*Imperata* *cylindrica*), and also an increase in abundance of white-leaved 'edelweis' (*Anaphalis longifolia*; Asteraceae) and bracken fern (*Pteridium aquilinum*) (Whitten *et al.* 1996). Recent *nuees ardentes* on Mount Merapi occurred in 2002, 2006 and 2010 (Sutomo and Fardila 2013; Sutomo *et al.* 2015), however MODIS only detected hotspots in 2002. MODIS also detected hotspots near Mt Rinjani and few of these burned years were matched with the years of eruption on Mount Rinjani (2004 and 2009). Fieldwork on Mount Rinjani conducted in 2014 confirmed that there were also fires that took place in the vast highlands of savanna which did not coincide with volcanic eruptions. Another example of this was the savannas around Mount Tambora. It is important to distinguish between detection of lava flows and eruptions to that where volcanic soils and regular fires (mainly in dry season) have maintained savanna. Many of areas with lots of fires are categorized as the latter (e.g. Rinjani and Tambora). In general, I have demonstrated the application of MODIS fire products to assist in detecting vegetation fires and fire scars in these areas ofJava, Bali, Lombok and Sumbawa Islands.

MODIS active fire detection product could detect lava if sufficiently hot or large relative to the surroundings (Roy, 2017, personal communication). However, MODIS active fire product not to considered to be a reliable lava detector (as it was not designed for this purpose), but it certainly does detect hot lava rather regularly (Giglio, 2017, personal communication). Landsat 7 enhanced thematic mapper plus (ETM+) has been used by several researchers to analyse the detailed thermal properties of active lava flows (Mouginis-Mark *et al.* 2000; Wright *et al.* 2002; Wright *et al.* 2001). MODIS active fire can detect thermal anomalies that, in terms of a volcano, usually comes in the form of hot volcanic plumes (Tassia 2016). Mount Merapi on Java Island is a classic example of a Merapi-type eruption which began with the collapse of lava dome and was followed by turbulence of extremely hot (up to 700°C) pyroclastic flows or *nuées* *ardentes* (Sutomo 2010; Sutomo and Fardila 2013; Takahashi and Tsujimoto 2000). In the case of Mt. Merapi, the pyroclastic flows burned vegetation in its path as it descended from the summit (Sutomo and Fardila 2013) and it is likely that the MODIS burn area product has detected the ash and coal remaining from the fires, especially when large enough (MODIS could not detect small fires). Therefore, I think the 'fires' detected by MODIS near Mt. Merapi in 2002 and 2006 were pyroclastic flows. Mount Merapi has been in one of the list of volcanoes that have been monitored by MODVOLC, a MODIS volcano detection algorithm (Wright *et al.* 2002).

The methodology used is likely to have underestimated the amount of fires in all land uses, but more likely in dense vegetation like forest. This may be because some fires are small and therefore not detected by MODIS burn area product. MODIS is also unable to sense fires through smoke or clouds. MODIS avoids this problem by looking for nearest cloudless day to detect change in cover, as well as the presence of charcoal. However, some months there are no such consecutive cloudless day periods available and so it can miss fires in this way. MODIS also does not detect small cool fires (NASA 2014). Further, much of the open vegetation outside of the widespread savannas of south-eastern Indonesia may not have sufficient fuel to carry fire, or are actively managed to exclude fires (especially likely in areas near human habitation). Fires are also formally excluded in most national parks and government applies strict regulations on agriculture and plantation burning. Another point to consider is the importance of peat fires in Indonesian fire story (Page *et al.* 2002; Putra *et al.* 2008). Peat fires in Indonesia may have been missed by the MODIS sensor. However the use of combination of MODIS with Landsat and disaster monitoring constellation (DMC) has been succesfully used to analyse burned area in a degraded tropical peat swamp in Central Kalimantan (Tansey *et al.* 2008).

This study results showed that although forests are burned at a lower rate than savannas, forests make up a slightly higher proportion of the total burned area. Nonetheless there is phenomenon of increasing fires in tropical forests. The changes in forest structure and composition was clear in an ENSO-induced fires in lowland tropical rain forest in east Kalimantan in 1997-1998 where fires had created 79% of open structure/gap in the forest (Simbolon *et al.* 2003a). Fire is not known or very rare in tropical rainforests and cool temperate forests/woodlands, and in other ecosystems with high and relatively consistent rainfall patterns, where litter and biomass mostly remain moist. Tropical rainforests are also noted for high rates of litter decomposition, which imply that there is little fuel accumulation. Humans have modified the structure of these dense, wet forests (by logging, clearing, road construction, etc.), which has resulted in exposure of ground fuels, as well as more plant debris on the ground, so that the fire risk in these vegetation types has increased. Consequently, large fires in rainforests are now more commonplace (Prasetyo *et al.* 2016; Siegert and Hoffmann 2000; van Etten 2010).

In many seasonally-dry places of East Nusa Tenggara (NTT), including on Sumba and Timor Islands, savanna is the dominant vegetation type. Regardless of their origins, most savanna is maintained by fire (Adejuwon and Adesina 1992; Archibald *et al.* 2005; Banfai and Bowman 2005; Cole 1960; Monk *et al.* 2000). Major sources of savanna fires, especially in the eastern parts of Indonesia where a dry climate persists, is more anthropogenic than natural (Fisher *et al.* 2006; Tacconi and Ruchiat 2006). High proportions of these fires as set intentionally for a range of reasons, including slash and burn agriculture and encouraging feed or clearing unwanted vegetation for cattle grazing (Fisher *et al.* 2006; Tacconi and Ruchiat 2006). Currently, local people in West Timor still practice land burning and shifting cultivation. Savanna fires are often started and promoted by local people for cultural and/or agricultural purposes (Fisher *et al.* 2006). I found fire detection over fourteen years on Timor Island to be similar to Sumba Island in terms of the number of hotspots.

Savanna in northern Australia share similarities with savanna in East Nusa Tenggara (NTT) in terms of temporal and spatial patterns of fires experienced and fire management practices, as well as invasive alien woody species (Beatty *et al.* 2015; Djoeroemana and Myers 2000; Radford *et al.* 2001a). Fire management issues affect the rural livelihoods of people in NTT Province in Indonesia (Beatty *et al.* 2015). People in this province rely heavily on subsistence dry land agriculture (Djoeroemana and Myers 2000). Although fire in NTT is extensively used by people in various agricultural, livestock management and hunting activities, uncontrolled fires are commonplace and incur considerable environmental and economic impacts. Similar to monsoonal eastern Indonesia, in northern Australia the major fire period occurs over the long dry season, typically between April/May–October/November (Russell-Smith *et al.* 2000a). People use fires for a variety of land management objectives and contemporary fire regimes are having disastrous impacts on native fire-sensitive species, communities, and habitats in some areas (Dyer *et al.* 2001). On the other hand, restricted application or absence of burning is in some cases leading to native and exotic woody species thickening/woody encroachment, equally with profound ecologic and economic cost (Belay *et al.* 2013; Radford *et al.* 2001a; Russell-Smith *et al.* 2000a). Increasing acknowledgment of these subjects has led to the development of joint fire management programs and co-operative approaches, linking practitioners from northern Australia working with relevant parties in eastern Indonesia and is expected to bring significant benefit to the study of regional to landscape-scale management issues (Beatty *et al.* 2015; Russell-Smith *et al.* 2000a).

El-Nino is a natural phenomenon occurring approximately once every four to five years where sea temperatures rise beyond the normal along the Pacific Equator. ENSO contributes to the temporal rainfall patterns in Indonesia and the strength and length of the monsoon season is believed to be interconnected to these fluctuations (Aldrian and Susanto 2003). Southern Oscillation Index or SOI is difference in mean air pressure across the Pacific Ocean used to measure ENSO. The SOI has been often used to explain the Indonesian rainfall variability with ENSO (Aldrian and Susanto 2003). Reports and study have reported a strong correlation between rainfall variations in Indonesia and the SOI (Aldrian *et al.* 2007; Aldrian and Dwi Susanto 2003; Hamada *et al.* 2002). Rainfall influences the moisture of potential "fire fuel" in a forest and or savanna ecosystem. More stable patterns of rainfall will reduce fire risk compared with fluctuating patterns (Brown and Davis 1973); this is likely because of fuel (e.g. grass) build up in wet years and then below-critical moisture levels in dry years. Although ENSO itself not a new phenomenon, the first time that many people in Southeast Asia heard about this climatic phenomenon was in 1997/1998 when severe forest fires occurred in east Kalimantan (Byron and Shepherd 1998). These fires had significant impact on the forest of Bukit Bangkirai, East Kalimantan, where Simbolon *et al.* (2003a) reported that the wildfires killed approximately 36 to 70% individual forest trees and reduced the basal area around 45 to 85% and the tree canopy from 79% to 23% cover. Other major forest fires as a result of prolonged dry season were recorded in 1982, 1983, 1991, 1994, 1997, 1998, and 2002 (1997-1998 and 2002 were El Nino years) (Sukmawati 2006). This study results showed that the drought period in 2012 (El Nino year) resulted in a series of fires which occurred on mountains across Java Island, from west to east. One of the mountains was in West Java, Mt. Papandayan in Garut Regency which was burned in August 2012 (Sutomo and Fardila 2012). Later in the year, Mt.

Agung in Bali Island was also burnt. This study has confirmed the findings of Fuller and Murphy (2006) that showed there is significant and strong correlation between ENSO period with fire events in Indonesia.

This preliminary analysis of spatial and temporal patterns of fires in Indonesia has demonstrated that medium to large-sized fires are more likely to occur in more open vegetation, particularly savanna, and such fires in the savanna vegetation are more common in eastern parts of the study area (where rain is generally lower and more seasonal). Additionally, in the higher rainfall parts of Indonesia, frequently burnt areas are associated with volcanic landscapes, some of which appears to be a long-term legacy of repeated burning following post-eruption recolonisation by grasses. More areas are burned during dry times linked to El-Nino episodes. Known savanna areas in the wetter parts of the study area (e.g. Baluran National Park in eastern Java) require regular burning to maintain open savanna vegetation which will transition to forest with fire exclusion (Sutomo *et al.* 2016). Much of the savanna and other open vegetation in the region are also active grazing lands (buffalo, cattle, native herbivores such as deer; Monk et al 1997) and so better understanding of the interactions between fire and grazing, and their impacts on productivity and biodiversity, should be a research priority. El-Nino forecasting could be important for fire planning in the region, given far more fires and land area is burnt during these periods.

Chapter 4: Structure and Composition of Seasonally Dry Tropical Forest of East Java, Indonesia, and Possible Transition to Savanna

ABSTRACT

Seasonally dry tropical forests (SDTF) and tropical savannas are important ecosystems which receive less research attention compared to tropical rain forest but equally are under serious threat. The objectives of this chapter are to characterize the structure and composition of SDTF and savanna of Baluran National Park, East Java, Indonesia, and to evaluate the possible evidence for transitions between SDTF and savanna. Fieldwork which sampled different sites with different fire histories in SDTF and savannas in Baluran using transects and plots were conducted in late 2014 and early 2015. Remote sensing image analysis was also employed by making use of the MODIS burn area product and various thematic maps. Results of this study suggest SDTF at Baluran has relatively high tree cover, low species diversity and a prominent vegetative response to fire. SDTF and savanna seems to be closely related in this region with long unburnt savanna potentially shifting to dry forest. The likelihood of ecosystem shifting is based on several lines of evidence: firstly, similarity in terms of species composition that shows the occurrence of forest tree species in savannas, secondly, similarity in terms of soil types and nutrients; and lastly spatial-temporal patterns of intentional burning and/or natural fires using remote sensing that shows areas where woody plants have displaced grasses in areas not burnt for at least 14 years. Knowledge on this ecological shifting is important for the ecosystem management, especially in terms of their usage by large native and exotic mammals.

Keywords: Seasonally tropical dry forest (SDTF), savanna, structure, composition, Baluran National Park, transition, remote sensing.

4.1. INTRODUCTION

Seasonally Dry Tropical Forests (SDTFs) and tropical savannas are important habitats which have received far less attention compared to tropical rain forests (Miles *et al.* 2006). Dry forest is understudied even though its conservation status is in some places more endangered than rainforest (Sunderland *et al.* 2015; Vieira and Scariot 2006). Although SDTFs are typically less diverse than tropical rain forests, they still harbour a wide variety of wildlife such as monkeys, deer, large cats, rodents and ground-dwelling birds. Seasonally Dry Tropical Forests (SDTFs) which are also known as tropical deciduous forests, monsoon forests or tropical dry forests – are forests of tropical regions that endure a prolonged dry season followed by a pronounced short period of heavy rainfall (Vargas *et al.* 2008). In such forests, many tree and shrub species shed their leaves during the dry season and will start forming leaves again early in the next rainy season (Justiniano and Fredericksen 2000). In many places, SDTFs tends to be bordered by or are mixed together with savanna ecosystems (Prance 2006; WWF 2012). Miles *et al.* (2006) presented a global distribution map of SDTF based on MODIS Vegetation Continous Fields (VCF) and reported that there are about 1,048,700 km^2 of dry tropical forest worldwide (about half of which occurs in South America) and that only 3 percent of this land is protected in conservation reserves. The MODIS data shows that only about 3.8 percent of the world's SDTFs are in Australia and Southeast Asia where they are at risk from human disturbance (Dale 2011; Miles *et al.* 2006).

A number of studies have also demonstrated that SDTF and savanna are under serious threat (Prance 2006). Miles *et al.* (2006) and Dale (2011) argued that 97% of SDTF globally is at risk of human disturbances and are highly vulnerable to destruction. Dirzo *et al*., (2011) shows that fires are threat and major modifier of this ecosystem. Approximately 75% of the original extent of SDTFs in Australia has been cleared and, despite most now being protected, these SDTFs continue to face a diverse range of threats which often work synergistically, such as climate change, fire damage, mining, livestock damage and weed infestation (Tng and Standish 2017). Large amounts of SDTF have been burned and cleared to establish farmlands due to their suitability for agriculture and very few studies have been devoted to examine their natural regeneration following such disturbances; in fact we know very little about the effects of fire in SDTF (Dirzo *et al.* 2011; Vieira and Scariot 2006). One of the natural mechanisms of dry forest regeneration, especially after fire, is through vegetative means such as resprouting (regrowth from dormant vegetative buds, typically under the bark or underground). Widespread resprouting gives rise to more resilient biomes that are able to cope with harsh disturbance regimes (e.g. regular and/or intense fire) (Clarke *et al.* 2013).

Although mostly confined to the neotropics (Dirzo *et al.* 2011), SDTFs are also widespread in sub-Saharan Africa and South Asia, but are relatively uncommon in Southeast Asia and Pacific regions, with the exception of the Indochina, the Sundas and northern Australia (Sunderland *et al.* 2015). In Indonesia, SDTF occurs in several locations, but are found predominantly in the eastern parts of archipelago. Miles *et al.* (2006) only recognized SDTF in the Lesser Sunda Islands of the Eastern Indonesia Province (Indonesian: Nusa Tenggara Timur/NTT). Studies are lacking for SDTF in Indonesia, especially when considering the large areas of SDTF and savanna in NTT (Beatty *et al.* 2015; Fisher *et al.* 2006; Monk *et al.* 2000; Russel-Smith *et al.* 2007; Tacconi *et al.* 2007). In addition to NTT, one of the locations in Indonesia where SDTF and savannas co-occur is Baluran National Park, East Java. Lowland dry forest in Baluran National Park (BNP) covers an area of *c.*1,500 ha. These forests are located in areas adjacent to savanna, teak forest and “evergreen block forest” (which refer to small blocks of dense forest *c.*2 ha in area that are green throughout the year, i.e. with no to few deciduous plants). Various animals including large nocturnal cats and other carnivores, such as the Javan leopard *Panthera pardus melas*, use these forests to gather their food during the cooler nights to conserve water. The combination of extensive areas of savanna and SDTF in Baluran National Park plays a significant role in supporting its diversity of plants and big mammals, including herbivores such as the Javan banteng (*Bos javanicus* subsp. *javanicus*), wild cattle found in Java (and Bali) Indonesia. Banteng is an endemic animal of Southeast Asia which is categorized as endangered on the IUCN Red List (IUCN 2008; Sabarno 2002; Suhadi 2009; Widayanti 2010). There are three subspecies of banteng recognised namely Javan banteng (*B. j. javanicus*), Bornean banteng (*B. j. lowi)*, from Borneo and Burma banteng (*B. j. birmanicus*) in Myanmar, Cambodia, Laos, Thailand and Vietnam (Timmins *et al.* 2008). As for Javan banteng, only seven groups of them with approximately 50 individuals each are known to remain in its native range (Caesariantika *et al.* 2011).

There is still considerable debate regarding the origins of the savanna (Monk *et al.* 2000), especially in Southeast Asia (Ratnam *et al.* 2016). Key questions are: are these savannas are derivative of SDTF and other tropical forests, or are they are formed naturally? Are they recently converted through fire or are they older systems which are reflective of soil/microclimate distinctions, with fire a secondary force? Research in Brazil, Africa and Australia showed that regardless of their origins, most savanna is maintained by fire, and will typically convert to woody-dominated vegetation in absence of fire (Adejuwon and Adesina 1992; Archibald *et al.* 2005; Banfai and Bowman 2005; Cole 1960; Monk *et al.* 2000). Currently there are three categories recognized regarding the origins of savannas, namely climatic savanna, edaphic savanna and derived savanna (Ford 2010; Murphy 2008; Scheiter 2008). The latter is to described savannas that are formed due anthropogenic disturbance that arise from forest clearing, burning, cattle grazing and other activities. In terms of a derived savanna, there is widespread opinion that savannas were originally dry forests/monsoon forests that are frequently burned (Hoffmann *et al.* 2012b; Monk *et al.* 2000; van Steenis 1972). For Indonesia, information on the effects of various disturbances to forest, and relationships between forest fire intensity and savanna community, is currently inadequate. Research into the origins of savannas in Indonesia is therefore needed. Savannas cannot be managed without knowledge and understanding of what factors drive its formation and what factors maintain its condition as a savanna.

Intermixing and coexistence of SDTF and savannas are also clearly seen in many regions for example in Mexico, and the Brazilian *cerrado* (Gardner 2006; Perez-Garcia and Meave 2006; Silva et al. 2006). Savannas and SDTF are often seen to form mosaics and a small distinction in factors such as soil, topography, disturbance type/history and micro-climate can resolve whether savanna or dry forest will arise in a particular area (de L. Dantas *et al.* 2013; Prance 2006; Rotter and Rebertus 2015). Due to this repeatedly close link shown by the two habitats it is valuable to study SDTF jointly with savannas. Therefore, the objectives of this chapter are to characterize the structure and composition of SDTF and savanna in Baluran National Park, East Java, Indonesia, and to evaluate the possible evidence for ecological transitions between these ecosystem types. This is the first study in this region that attempts to link seasonal dry tropical forest of Baluran National Park with the formation of the savanna ecosystem, which also is a significant feature of the national park and one of the reasons it is protected.

4.2. METHOD

4.2.1. Study Site

Baluran National Park (BNP) is located in Situbondo Regency, East Java Province. The park is bordered by the Madura Strait to the north and Bali Strait to the east. The park covers an area of about 25,000 ha, located between 7° 29' 10" and 7° 55' 55" S and 114° 29' 20" and 114° 39'10" E, and characterized by a single volcanic cone of Baluran Mountain having a summit at 1,247 m (Tjitrosoedirdjo *et al.* 2013). Baluran has monsoonal type of climate affected by dry easterly winds, particular in the dry season (Sabarno 2002). Precipitation in Baluran National Park ranges from 900 - 1,600 mm/year with average nine months of no or low rainfall (generally April to November). Highest rainfall usually occurs in December up to February. Baluran Mountain provides a rain shadow so that rainfall is lower and more seasonal than other parts of East Java.

4.2.2. Data Collection and Analysis

A). Structure and Composition SDTF

Field data was collected in Baluran National Park during January (wet season) and August 2015 (dry season). The fire history of the study area was acquired from various sources of unpublished reports, interviews with park managers and rangers, and from the MODIS burned area product data for years 2000 – 2014. For investigating structure and composition of SDTF, sampling was conducted in lowland SDTF on eastern side of Mt. Baluran in a north-south transect following the road (Figure 1). This area was chosen because of the large areas of continuous forest with easy access. Based on the fire history data, sampling was conducted in: recently burned SDTF (last burnt 2012 to 2014), 10 years since last burned SDTF (2004 to 2006) and long unburned SDTF (no evidence of fires for at least 11 years). Within forest patches of a certain fire age, a plot 50 x 50 m in size was randomly placed to sample species composition and abundance of tree and groundcovers layers. In total there were 15 plots established. Within each plot, each species was identified, and the height and diameter of all mature trees (with bole dbh ≥ 20 cm), pole trees (pole dbh 10-20 cm) and saplings (dbh 2-10 cm) were recorded. The number of understorey plants and seedlings was also counted (Kent 2011). Plant identification was assisted by taxonomists at the Bali Botanical Garden, and also following nomenclature of Backer & van den Brink (1963), and van Steenis (1972) and then names updated to reflect "The Plant List" (ThePlantList 2010). I also sampled fuel load/biomass/leaf litter data in small 1 x 1 m quadrates within each plot. In recently burned SDTF, I also recorded which species re-sprouted after fire, the number of plants resprouting, and the proportion of basal to above-ground resprouting within the plots.

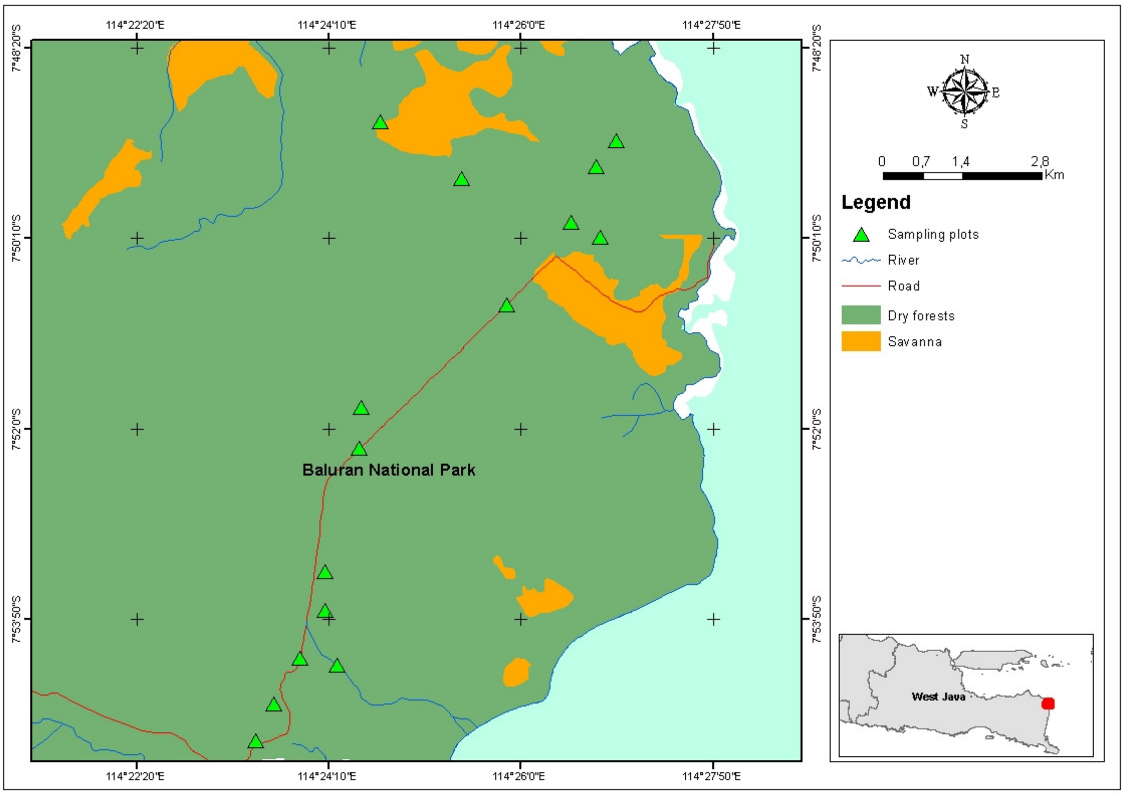


Figure 1. Sampling Plots in Seasonal Tropical Dry Forest of Baluran National Park East Java Indonesia

Vegetation analysis was then conducted to calculate and compare the values of Importance Index (IVI) for each habitat/stratum, and to compare species richness and diversity of trees and groundcovers. Importance Value Index for each stratum (trees, poles, sapling, seedling and groundcovers) was calculated using the formula from Kent (2011). Importance Value Index was calculated for each species of tree and groundcover in each of the study sites (plots) and then averaged for each time since last fire class. Ordination analysis was conducted using Non-metric Multidimensional Scaling (NMDS) (on the site X site dissimilarity matrix based on Bray-Curtis measure and cover values) in the PRIMER v.6 Software (Clarke and Gorley 2005). SIMPER (Similarity Percentage) analysis was then used to explore the relative contribution of individual species to dissimilarity among SDTF plots, also within PRIMER v.6. Species responses to fire (resprout, basal and above ground) were graphed in Excel.

B). Supports for Transitions

In order to search for evidences of ecosystem transition, I compared species composition in SDTF and savanna. I chose 11 locations of relatively long unburned savanna (ten years of fire absence), 11 locations of relatively unburned SDTF (five years of fire absence), 12 recent burned SDTF (1 year) and 10 of recently burned savanna which are burned at least twice. In these sites, plot of 50 x 50 m in size were laid out to sample species composition and abundance of tree and groundcovers layers. Soil samples were also collected from dry forest and savanna plots. The soil samples were then taken to the Soil Laboratory at Udayana University Bali for analysis of pH, soil organic matter and nitrogen. Square-root transformation of vegetation data was used to construct resemblance matrices based on Bray-Curtis similarity index. These matrices formed as the basis for NMDS (with time since fire as overlay vector) and cluster analysis to compare species compositions in the savanna with the SDTF. SIMPER

(Similarity Percentage) analysis was then used to explore the relative contribution of individual species to dissimilarity among savanna and SDTF plots. This multivariate analysis makes use of the PRIMER v.6 package.

I also conducted remote sensing analysis using the standard overlay method of the following data: MODIS burned area product 2000-2013 downloaded from the website http://modis.gsfc.nasa.gov/.; land use map for the years 2000 and 2013 for BNP obtained from the park management office (this is a map of the main ecosystem types for management purposes – it is derived based on field observations combined with aerial photo interpretation), and 2013 Google Earth images. MODIS burn area data was overlaid with the BNP land use map. From this analysis I extracted information on which locations were burned in the range period of 2000-2013 in BNP. I also calculate how much areas were burned, and the average peak time of the year when fires occurred. To check whether there is a change in landuse (savanna to SDTF or vice versa) I overlaid the BNP landuse type map (for year 2000; Figure 2) with the current (2013) BNP landuse type map and the MODIS fire mapping (2000-2013). This remote sensing analysis was done using ARC-GIS v.10.1.

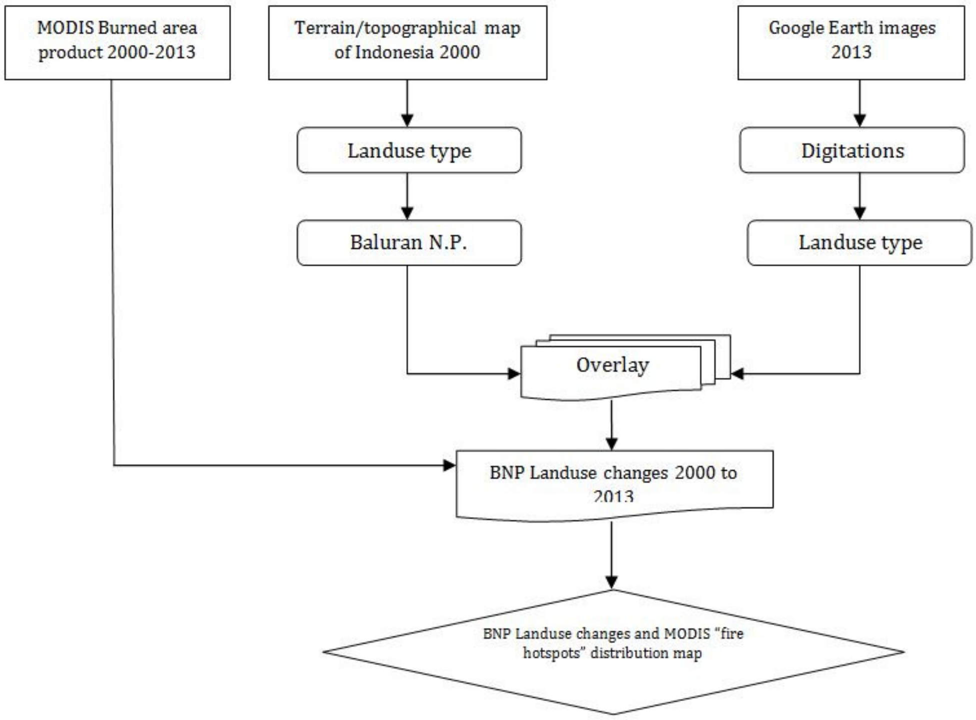


Figure 2. Flowchart diagram of remote sensing work for detecting land use change which may relate to fires in Baluran National Park.

4.3. RESULTS

4.3.1. Structure and Composition of SDTF

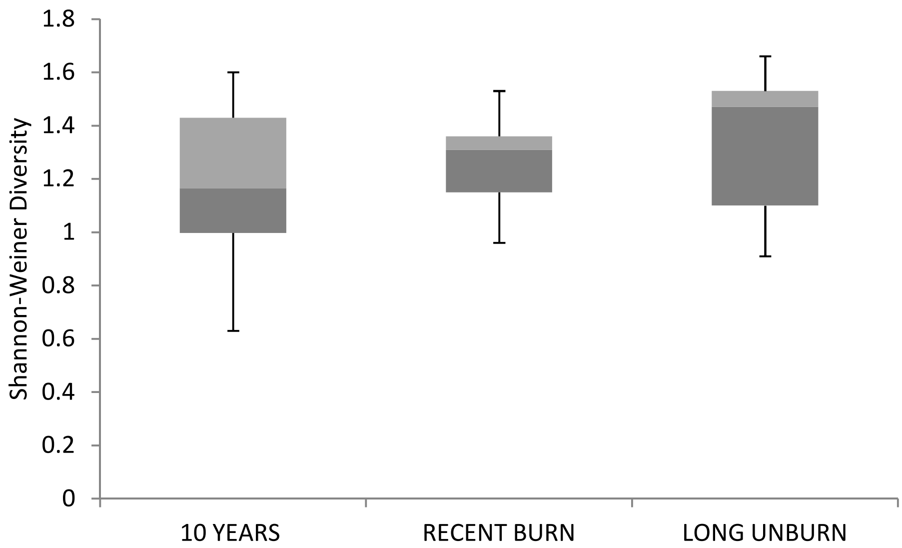
A total of 20 species in 19 genera and 13 families were recorded in the SDTF. In the mature trees strata, there were eight species (in eight families) that shows IVs greater than 10 (Table 1). The highest IV (75.14) belonged to *Glochidion sumatranum* (Phyllanthaceae), whereas the lowest (11.05) was *Flacourtia rukam* (Salicaceae). Similarly, in the young tree (poles) stratum, there were eight species (but in seven families) with *Streblus asper* (Moraceae) having the highest IV (72.96) and *Acronychia trifoliata* (Rutaceae) the lowest (11.65). In the sapling strata, ten species belonging to seven families have IVs of greater than 10 being the same species in the tree strata, with *G. sumatranum* being the highest (55.39) and *Tamarindus indicus* (Fabaceae) the lowest (12.94). Notably in this stratum is the appearance of exotic invasive species *Acacia* *nilotica* (*Vachellia nilotica*) with an IVI about half that of the *G. sumatranum*. Groundcover strata comprises of five species in three families: *Brachiaria ramosa* (Poaceae,) has the highest IV (22.8), whereas *Cosmos caudatus* (Asteraceae, annual herb) had the lowest (10.32).

Table 1. Result from Importance Value Index calculation (IVI for mature tree, young trees (poles), saplings and groundcover stratum in lowland SDTF Baluran N.P.

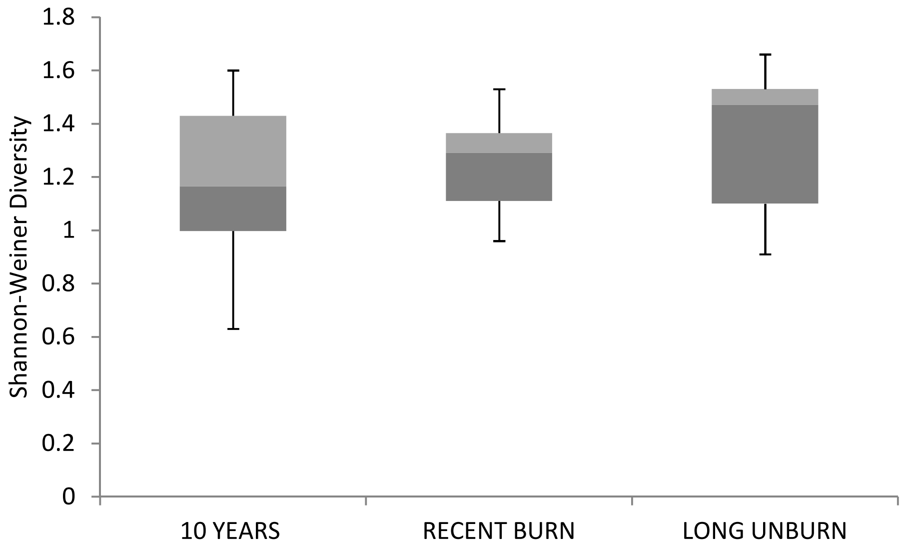
Species / Stratum	**Family**	**Habitus**	**Importance Value Index**
	Mature Tree (dbh ≥20 cm)		
Glochidion sumatranum	Phyllanthaceae	Tree	75.14
Premna tomentosa	Lamiaceae	Tree	51.35
Schoutenia ovata	Tilliaceae	Tree	28.76
Tamarindus indicus	Fabaceae	Tree	25.34
Schleichera oleosa	Sapindaceae	Tree	25.25
Flacourtia rukam	Salicaceae	Tree	15.73
Streblus asper	Moraceae	Tree	14.65
	Pole (dbh 10 – 20 cm)		
Streblus asper	Moraceae	Tree	72.96
Capparis sepiaria	Capparaceae	Tree	57.30
Schoutenia ovata	Tilliaceae	Tree	44.89
Glochidion sumatranum	Phyllanthaceae	Tree	43.92
Antidesma bunius	Phyllanthaceae	Tree	34.21
Premna tomentosa	Lamiaceae	Tree	13.43
Kleinhovia hospita	Malvaceae	Tree	12.21
Acronychia trifoliata	Rutaceae	Tree	11.65
	Sapling (height > 1.5 m &dbh< 10 cm)		
Glochidion sumatranum	Phyllanthaceae	Tree	55.39
Capparis sepiaria	Capparaceae	Tree	48.94
Schoutenia ovata	Tilliaceae	Tree	42.14
Streblus asper	Moraceae	Tree	28.41
Acacia nilotica	Fabaceae	Small Tree	25.38
Antidesma bunius	Phyllanthaceae	Tree	23.04

Kleinhovia hospita	Malvaceae	Tree	20.05
Abutilon indicum	Malvaceae	Small shrub	15.72
Hibiscus panduriformis	Malvaceae	Shrub	13.46
Tamarindus indicus	Fabaceae	Tree	12.94
	Groundcovers		
Brachiraria ramosa	Poaceae	Grass	22.80
Schlerachne punctata	Poaceae	Grass	22.25
Mimosa invisa	Mimosaceae	Herb	18.64
Oplismenus burmannii	Poaceae	Grass	12.80
Cosmos caudatus	Asteraceae	Annual Herb	10.32

Shannon-Wiener species diversity showed no significant differences between TSF (time since fire) classes (F = 0.53, P > 0.05). The diversity index shows low to moderate species diversity levels. Tree species diversity is highest (1.43) in the long unburned forest compared to recently burned and 10 years since last burned forests (Figure 3). Recently burned and 10 years since last burned forests had almost the same tree species diversity (0.76 and 0.78). Groundcovers species diversity generally shows moderate levels (1.17 to 1.33) in all studied forests locations.



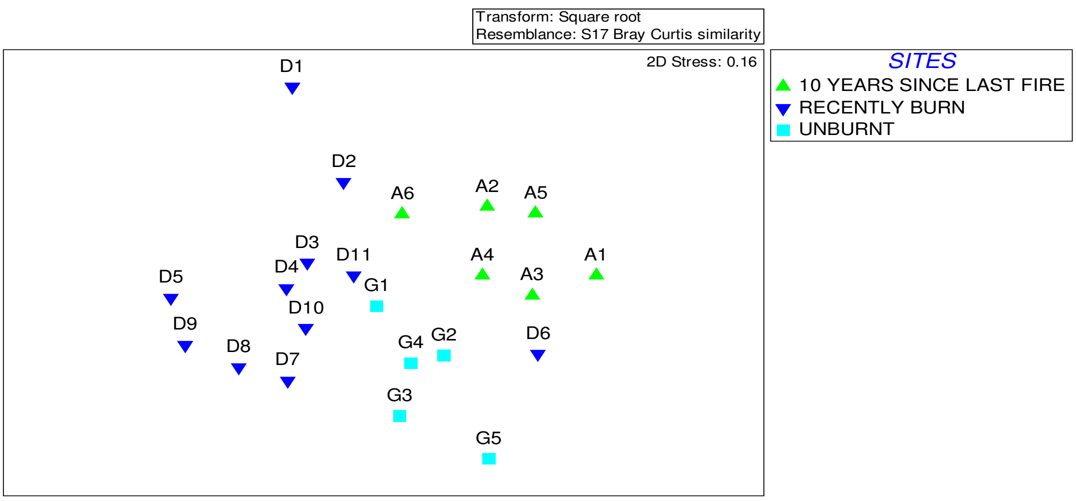
A



B

Figure 3. Box plot of tree species diversity (A) and groundcover diversity (B) among sites with different fire age and one long unburned in dry forest of Baluran National Park.

Overall non-metric multidimensional scaling (NMDS) of the Bray-Curtis species similarity of all vegetation plots of the three dry forests (recently burned, 10 years since last burned and long unburned) showed a clear separation between them (Figure 4). There were significant differences in species composition between fire age classes with a global ANOSIM *R* value of 0.4 (for tree species) and 0.6 (for groundcover species) ($P <$ 0.1%).



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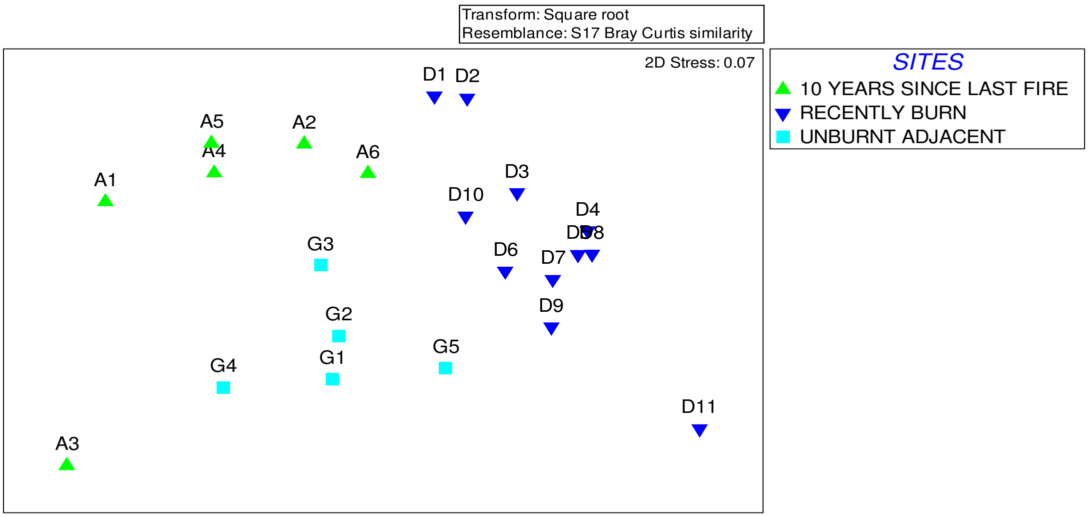


B

Figure 4. Two dimensional NMDS ordinations of sampled sites (10 years since last burned, recently burned and unburned) based on Bray-Curtis similarity (square-root transformation). (A): NMDS graph based on tree species composition. Global *R* ANOSIM = 0.4. (B): NMDS graph based on groundcover species. Global *R* ANOSIM = 0.6

Results of the similarity percentage analysis (SIMPER) showed which species were most important in differentiating between the groups (Table 2). In group 10 years vs. recent fire for the groundcover strata, *Sclerachne punctata* (grass) is the species that the most important discriminating species, then followed by *Oplismenus burmannii* (grass), *B. ramosa* (grass), and *Mimosa invisa* (Fabaceae). For tree strata, *Premna tomentosa* was the most important species differentiating the 10 years TSF group from the recently burned dry forests, followed by *G. sumatranum* and *Schoutenia ovata*. There were no species which consistently discriminated (dissimilarity/SD >2) between '10 years' and 'recent fire'. For groundcover strata, in TSF groups 10 years vs. unburned and recent vs. unburned, *Cosmos caudatus* was the most important differentiating species (Table 2). For tree strata, in group 10 years vs. unburned, there were six species that differentiated the dry forests with *F. rukam* was the most important species as well as the more consistent. Whereas in group recent fire vs. unburned, five species differentiate and *P. tomentosa* was the most important differentiating species however it was not the most consistent species, but *F. rukam* was (Table 2).

Table 2. Similarity Percentage (SIMPER) analysis results. Species percentage contribution to the dissimilarity between the three dry forests group (Recent burned, 10 years since last burned and long unburned). Larger values of the ratio of average Dissimilarity to SD indicates that not only does that species contribute much to the dissimilarity between dry forests but that it also does consistently in inter-comparisons of all samples in the three dry forests. Diss/SD > 2 indicates species which strongly discriminate between groups.

	Diss/SD			**Contrib%**		
Species	**10 years vs recent**	**10 years vs unburnt**	**recent vs unburnt**	**10 years vs recent**	**10 years vs unburnt**	**recent vs unburnt**
			Groundcovers			
Brachiaria ramosa	1.27	-	-	10.03	-	-
Cosmos caudatus	-	1.7	1.8	-	16.36	12.83
Oplismenus burmanii	1.43	1.35		11.2	11.97	-
Schlerachne punctata	1.44	-	1.52	12.63	-	10.41
Mimosa invisa	0.88	0.58	0.98	8.32	3.51	8.03
			Trees			
Flacourtia rukam	-	1.85	1.81	-	13.45	14.58
Glochidion sumatranum	1.44	1.30	1.25	21.03	9.31	10.03
Premna tomentosa	1.4	1.07	1.45	21.28	7.5	44.11
Schleichera oleosa	-	0.92	0.93	-	13.94	15.04
Schoutenia ovata	0.82	0.89	1.04	11.35	9.17	9.67
Streblus asper	-	0.69	-	-	9.37	-

In terms of fuel load, results showed that there is a modest positive correlation between fuel load and time since last fire (Figure 5).

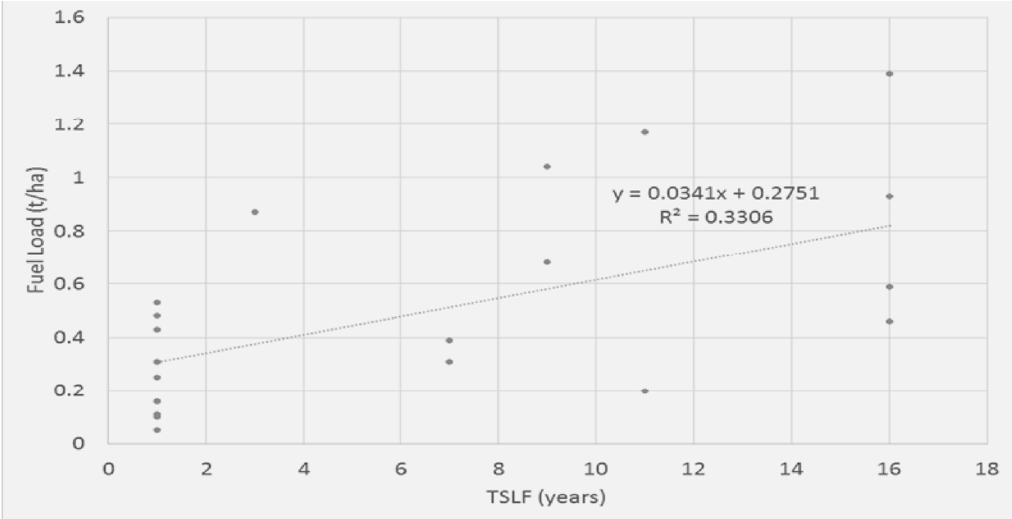


Figure 5. Correlation fuel load with time since last fire in SDTF Baluran National Park.

Two types of resprouting were found in recently burned plots: basal and above ground, with the number of individuals with above-ground resprouting being higher than number with basal resprouting (Figure 6).

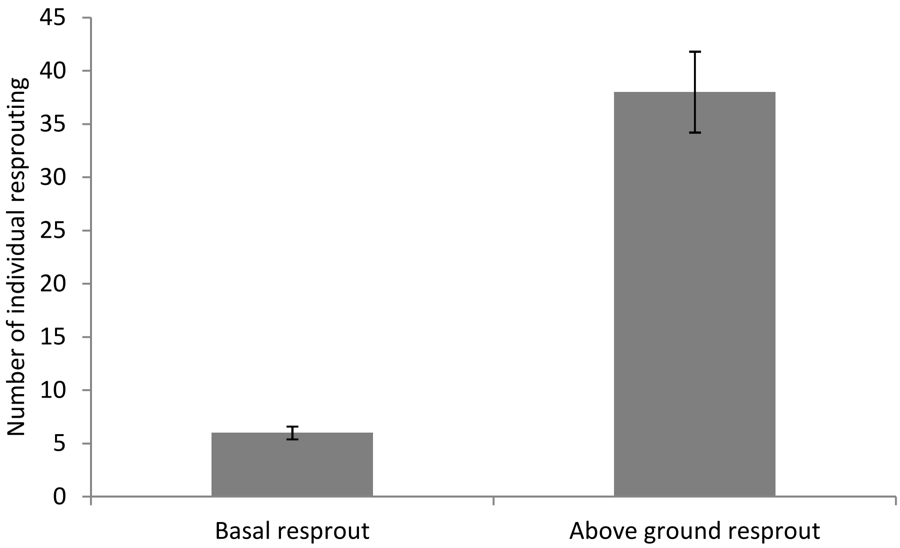


Figure 6. Mean number of individuals (per plot) resprouting at recently burned localities for two main types of resprouting in Baluran National Park.

A different set of species, namely *P. tomentosa, G. sumatranum*, *Schoutenia ovata*, *F. rukam*, *Acacia nilotica*, *Microcos tomentosa*, *Bauhinia* sp., and *Acacia tomentosa,* were observed resprouting following recent fires (Figure 7). *Premna tomentosa* shows the highest number of individuals resprouting after fire, whereas the lowest was *A. tomentosa*. Three members of the Fabaceae family show resprouting ability (*Acacia* and *Bauhinia*), including the exotic invasive species *Acacia nilotica*. In terms of resprouting percentage, *A. tomentosa, Bauhinia* sp., *F. rukam*, and *M. tomentosa* have the highest percentage of resprouting individuals followed by *G. sumatranum* (91%), *P. tomentosa* (77%), *A. nilotica* (75%) and, the lowest *S. ovata* (57%) (Figure 8).

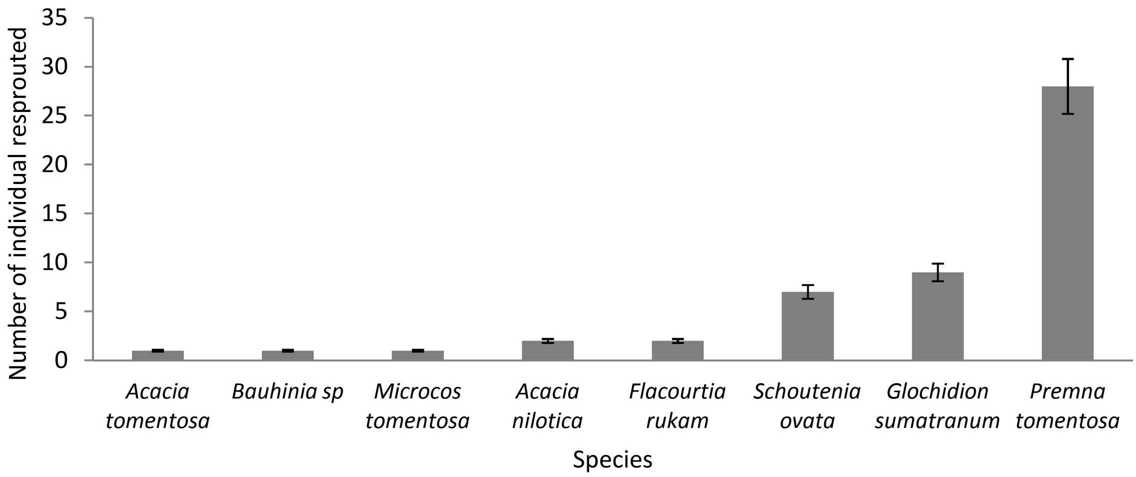


Figure 7. Histogram illustrating mean number of individuals (per plot) of selected major species that resprouted following recent fires in Baluran National Park's SDTF.

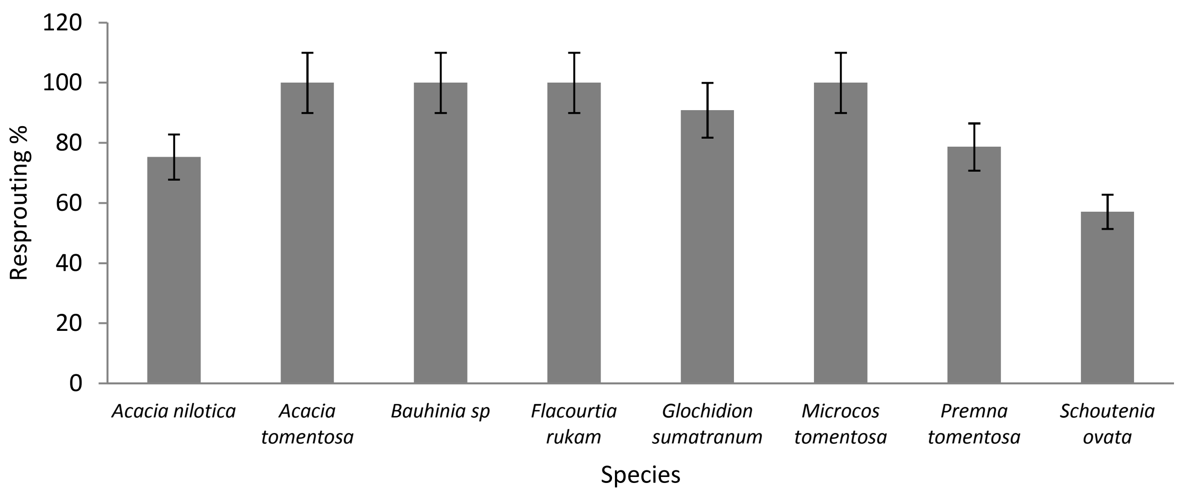


Figure 8. Histogram showing percentage of resprouting individuals of selected major species in SDTF of Baluran National Park following recent fire.

4.3.2. Supports for Transition

Overall the non-metric multidimensional scaling (NMDS) ordination result shows the plant communities are moderately distinct (Global *R* ANOSIM = 0.53, *P* < 0.1%). There is separation of burned and unburned sites along the vertical axis, and of dry forest and savanna sites along the horizontal axis. However quite a few of the recently burned SDTF sites are intermediate between burned savanna and long-unburned SDTF. Also some of the long-unburned savannas are grouped with or close to long-unburned SDTF (Figure 9). Site age (year/time since fire) had high correlation with species composition (P = 0.001 and *rho* = 0.702; Figure 9). Older SDTF and savanna sites (SDTF long unburned and savanna long unburned) tend to cluster to the bottom left-hand side of the ordination which is aligned to the time since fire (Spearman) correlation vector (Figure 9).

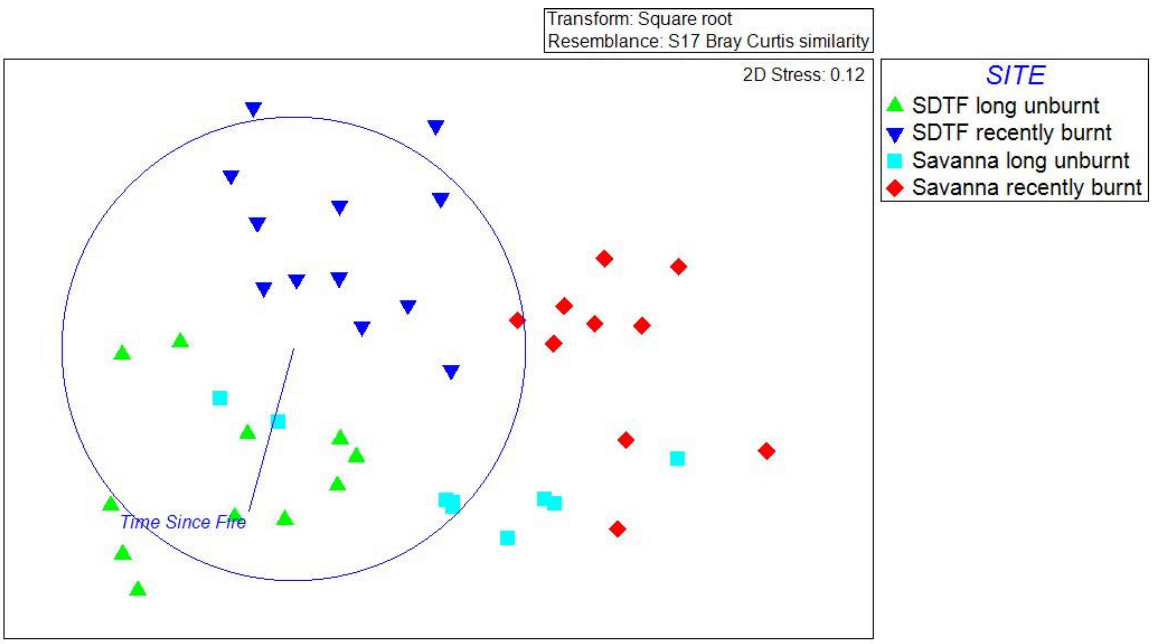


Figure 9. NMDS analysis result for SDTFs and savannas sites with different time since last fire class in Baluran National Park. Global *R* ANOSIM = 0.53, *P* < 0.1%. The Spearman correlation of time since fire has also been included as a vector displaying the direction and influence on distribution of sites. Spearman correlation value on MDS axis 2 = 0.702.

Cluster analysis revealed there is a relatively moderate to high percentage of similarity in terms of species composition between several sites of unburned SDTF group with sites of long unburned savanna group (54.55%, 66.67% and 72.73%) (Figure 10). Pairwise test of the two groups showed they were not different with a relatively low R value of 0.35 (P > 0.1%).

SIMPER analysis result shows ten tree species that mostly contributed to differentiating between the SDTF and savanna (Table 3). *Erythrina euodiphylla* and *F. rukam* were species that contributed the most (12.66% and 11.25%) in the SIMPER result. *E. euodiphylla* and *F. rukam* were both present in the two groups but with different abundance. *E. euodiphylla* was more abundant in SDTF whereas *F. rukam* was more abundant in the long unburned savanna. In total, there were seven tree species that were present in both groups. There was only one species that were absent in SDTF and only present in the long unburned savanna, namely *Acacia leucophlea*. In the opposite, there were five species that were absent in the savanna and only occur in SDTF. Species that normally found in SDTF in Baluran National Park such as walikukun (*S. ovata*), rukem (*F. rukam*) and asam jawa (*Tamarindus indicus*) were also found in the long unburned savanna site. Complete species list in this chapter, which indicate whether species are typical of dry, rain forest or savanna and also information on native/naturalized species can be found in Appendix 2.

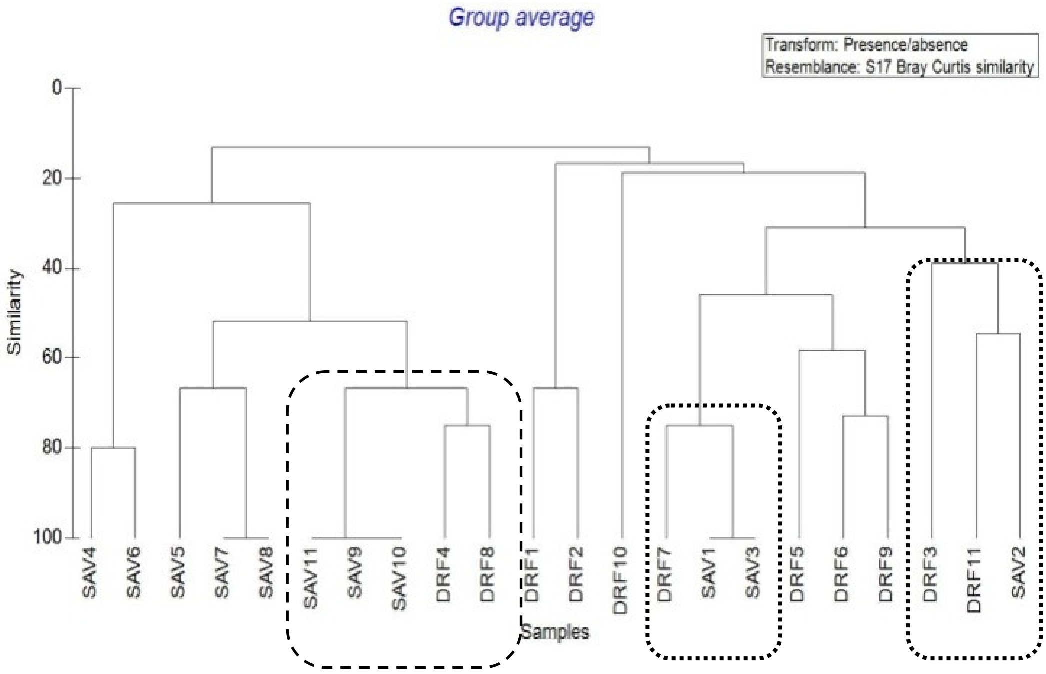


Figure 10. Dendrogram cluster analysis of community composition in long unburned savanna (SAV) and unburned SDTF (DRF), Baluran National Park. R = 0.35, $P < 0.1\%$.

Table 3. SIMPER analysis result: Percentage (%) contribution of species to average Bray–Curtis dissimilarities in all pairs of sites. Only those species with a contribution to average dissimilarity of >5% are included. Species in bold refers to species that is commonly found in SDTF in Java Island (Whitten *et al.* 1996).

	Unburned SDTF	**Unburned savanna**		
Species	**Av.Abund**	**Av.Abund**	**Contrib%**	**Cum.%**
Erythrina euodiphylla	1.87	0.47	21.15	21.15
Flacourtia rukam	0.33	1.28	11.25	32.4
Hibiscus panduriformis	0.82	0	8.71	41.11
Schoutenia ovata	2.55	2.46	8.54	49.65
Schleichera oleosa	0.47	0.67	8.26	57.91
Grewia eriocarpa	0.67	0	7.79	65.7
Acacia leucophlea	0	0.67	6.51	72.2
Capparis sepiaria	0.33	0.33	5.15	77.35
Tamarindus indicus	0.33	0.33	4.92	82.27
Glochidion sumatranum	0.33	0	3.81	86.08
Cordia bantamensis	0.33	0	3.56	89.63

Helicteres isora	0.33	0	3.56	93.19

The result from soil analysis show not much soil difference between SDTFs and savannas in Baluran National Park (Table 4).

Table 4. Soil sample analysis result for savanna and SDTF in Baluran National Park

Location	**Soil**	**pH**	**C %**	**N %**
Savanna	Volcanic	6.74 Neutral	3.51 High	0.23 Medium
SDTF	Volcanic	6.65 Neutral	4.39 High	0.27 Medium

Transitions from savanna to dry forest in Baluran National Park based on land use mapping are shown in figure 11. As much as 280 ha of the savanna shifted to dry forest in the 14 year period 2000-2013. Most of the shifted area experienced little or no fires (at least as detected by MODIS) during the same period. Conversely, shrubland with frequent fires were observed to have also shifted to savanna during the similar period. There were 1,711 fire "hotspots" detected in Baluran National Park across this period. Fire detection was higher in the dry months of June, July, August and September, with August being the month which usually fires were detected the most. Seasonally dry tropical forest burnt the most (2,629.6 ha), then followed by savanna (1,401.4 ha), primary forest (600.9 ha) and savanna invaded by *Acacia nilotica* (394.9 ha).

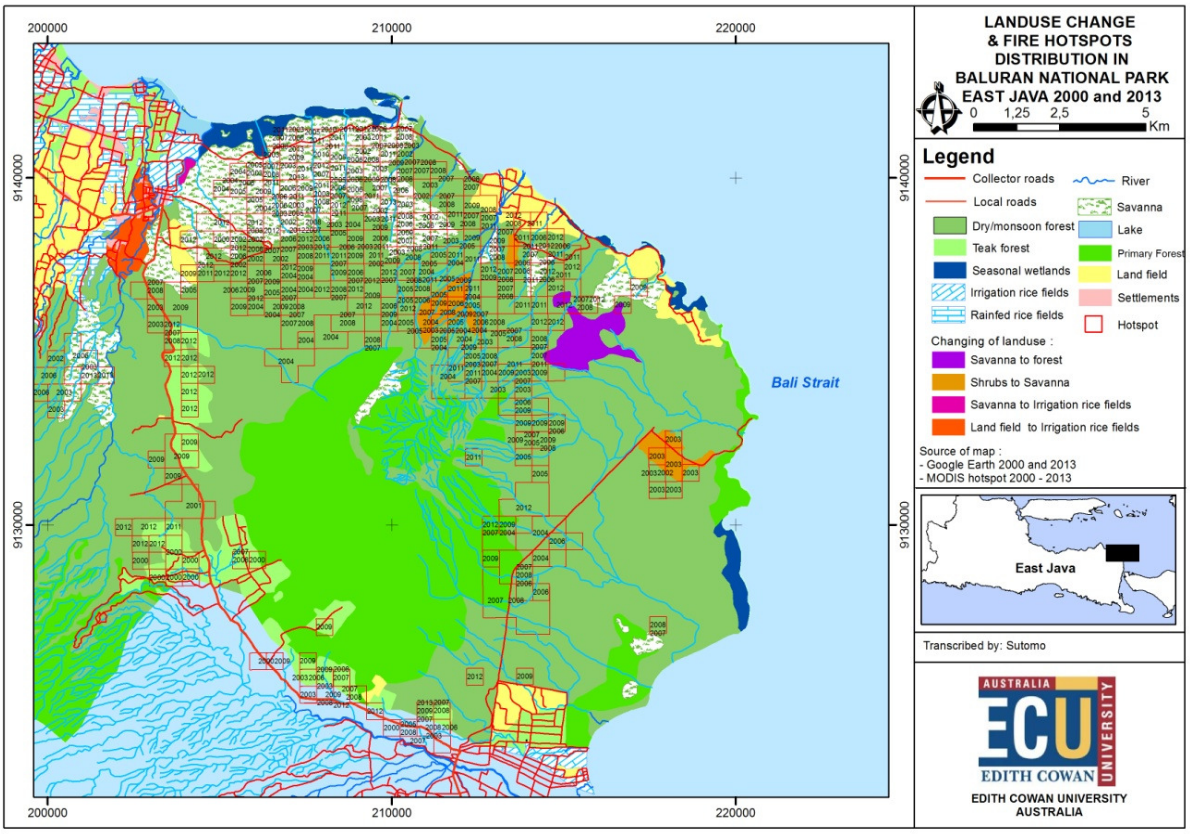


Figure 11. Landuse type changes in Baluran National Park and MODIS hotspots distribution in year 2000-2013. Land field or known by the local as "Tegalan" is land usually utilize by local people to plant food crop such as cassava, corn or fruit trees. Note the change in savanna to dry forest (marks in purple colour area) covers an area of 280.39 ha.

4.4. DISCUSSION

Seasonally Dry Tropical Forest (SDTF) in Baluran National Park (BNP) have high cover as expected for forest but lower species diversity than wet (rain) forest. This is supported by field measurements across dry forests with different time since last fire in Baluran National Park. SDTF in Baluran National Park comprise of up to four strata: trees, poles (young trees), saplings and groundcover consisting of dicot herbs and grasses. The tree stratum is characterized by *P. tomentosa*, *F. rukam*, *S. ovata* and *T. indicus* (originally from Africa but now naturalized in Indonesia), confirming results by Whitten *et al.* (1996). A notable species present in the sapling stratum is the noxious exotic invasive species *Acacia nilotica*. *Acacia nilotica* was first planted in the 1960's in and around BNP as fire breaks, however it has now become widespread and is forming a dense thicket of pure *A. nilotica*, especially in Bekol area where it was observed to develop adjacent to savanna on one side and SDTF on the other. The presence of this species as saplings and seedlings in SDTF may be indicating that the species is starting to spread deep into the SDTF of Baluran National Park. There have been few reports on the expansion of *A. nilotica* stands into SDTF, unlike that reported in savanna and grassland in many regions (Radford *et al.* 2001b; Setiabudi *et al.* 2013; Smith and Goodman 1987). However, Kriticos *et al.,* (2003) argued that under current climatic conditions it is predicted that SDTF are at risk of invasion and expansion by *Acacia nilotica* will increase markedly. Expansion of *A. nilotica* into adjacent savanna in Baluran National Park is elaborated by Sutomo *et al*., (2016) and also in chapter 5 of this thesis.

SDTF is not as rich in species when compared to rainforests, but still retains a considerable amount of diversity (Blackie *et al.* 2014; Gonzalez-Rivas *et al.* 2006; Sunderland *et al.* 2015). I found Shannon-Wiener species diversity measures showed low to moderate species diversity in this type of forest with a total of 20 species in 19 genera and 13 families recorded across the 15 plots (total area sampled of 0.6 ha). Other studies report higher species diversity in SDTF, such as Gonzales-Rivas *et al*., (2006) in Nicaragua who found 29 families, 49 genera and 59 species of plants in 2 ha of SDTF surveyed. Although Dirzo *et al*. (2011) state there is not enough collated data to compare species diversity of SDTF across different regions of the globe, it appears alpha diversity of neotropical SDTFs are typically much higher than I have found in eastern Indonesia. However, I sampled in the early wet season and this may have influence my results. Also, I found no succulent species in Baluran SDTF which supports Pennington *et al*,'s (2009) proposition that this is one of the major distinctions between Asian and neotropical SDTFs (the latter having many succulent plants).

In the SDTF of Baluran National Park, resprouting is widespread, with the majority of its tree species consistently responding by resprouting following fires, with above-ground resprouting being the most common type. Vegetative regeneration is prominent in SDTF, especially for species that regenerate poorly from seed. In tropical dry forest in Bolivia, vegetative regeneration is one of the most important traits for regeneration following disturbance by logging (Stoner and Sanchez-Azofeifa 2009). The implication is that there will be rapid recovery following fire in SDTF but questions remain of the effect of repeated fires. However, if any species did not resprout but were killed, they could take a long time to recover and would facilitate transition to savanna vegetation. The ability of species to resprout following defoliation differs with age, size and fire severity (Hobbs and Mooney 1985). Relative few tropical tree species have the capacity to resprout when they are young, the exception being many leguminous savanna trees (Bond and Wilgen 1996), which was the case for three species in this study, including the exotic invasive species *A. nilotica.* Tropical Myrtaceae resprouting was observed in northern Australia. Williams *et al.* (1999) and Prior *et al.* (2010) studied eucalypt resprouting after fire in a savanna landscape.

Even though dry forests have species either resistant to fire or are advantaged from it, recurrent fires can simplify community species composition of SDTF (Vieira and Scariot 2006). In this study, SDTF species composition was found to differ between recently burnt plots and plots that have not been burnt for significant amount of time, suggesting the effect of fire on the community composition. Frequent fires may even cause a shift in the ecosystem as shown in our mapping results. It can be seen in the map that transition occurred from SDTF to savanna at specific locations where fires are a recurrent event. Fuel load is then an essential part of this system. My result shows that biomass/fuel load was highest at the long unburnt STDF and was lowest in the recently burnt STDF. Woody species biomass and leaf litter are known to reduce when frequent, high intensity fires occur (Bond and Wilgen 1996; Shannon *et al.* 2011).

Seasonally Dry Tropical Forest (SDTF) and savanna in Baluran seem to be closely related and indeed long unburnt savanna tends to shift towards SDTF in terms of species composition. The likelihood of ecosystem shifting is based on several lines of evidence: firstly, similarity in terms of species composition that shows the occurrence of forest tree species in long unburnt savannas;, secondly, similarity in terms of their soil types and nutrients; and lastly, spatial-temporal patterns of intentional burning and/or natural fires using remote sensing that shows areas where woody plants have displaced grasses in areas not burnt for at least 14 years. Further, my NMDS results shows that quite a few of the recently burnt SDTF sites are intermediate between burnt savanna and long unburnt SDTF which indicates a floristic shift towards savanna composition with burning. Also some of the long unburnt SDTF are grouped with or close to long unburnt savanna which also indicates a possible shift in species composition.

Savanna and dry forests are often considered as alternative states within a state-and-transition framework (Gillson and Ekblom 2009; Twidwell *et al.* 2013). Previous findings and evidence around the globe suggest that many savannas are not in a stable state, where climate, fire, and soil nutrient stocks are coupled, creating a dynamic coexistence between the savanna and forest (de L. Dantas *et al.* 2013; Hirota *et al.* 2011; Hoffmann *et al.* 2012a; Silva *et al.* 2013; Silva 2014). Currently there are three categories regarding origins of savannas namely climatic savanna, edaphic savanna and derived savanna (Ford 2010; Murphy 2008; Scheiter 2008). The latter is to described savannas that are formed due anthropogenic disturbance that arise from forest clearing, burning, cattle grazing and other activities. In terms of a derived savanna, several ecological reviews have concluded that certain savannas were derived from SDTF which were frequently burned (Hoffmann *et al.* 2012a; Monk *et al.* 2000; van Steenis 1972). Murphy and Lugo (1986) also considered savannas to be resulting from disturbed dry forest. My results imply that savanna in Baluran National Park is transitional, supporting the works of these researchers. Most of the transformed savanna areas experienced little or no fires detected during the 2000 – 2013 period. Conversely, areas of dense shrubs (these being *A. nilotica* invasion areas which have been managed/controlled) were observed to have shifted to savanna with frequent fires during the same period. However, the assessment on the role of fire in this transition is very often complicated by other effects of human-induced disturbances and also grazing (Murphy and Lugo 1986). Indeed, from my results, the area of dense shrubland showing signs of transition to savanna, besides experiencing frequent fire, apparently also endures heavy grazing from cattle owned by local people that live nearby this area of the Baluran National Park. As a consequent of the frequent burning and grazing pressure, usually the border regions or the boundary of dry forest and savanna is characterized by mosaic of forest patches, grassland, savanna, farms and fallow (Adejuwon and Adesina 1992; Murphy and Lugo 1986; Stuart *et al.* 2006).

The concept of alternative stable state/transition has been shown to be relevant when studying the dynamics of savanna ecosystems (Perrings and Walker 1997; Rietkerk *et al.* 2002). Shifting of one ecosystem to the other also found elsewhere (Archer 1991; Burrows *et al.* 1986; de L. Dantas *et al.* 2013; Silva *et al.* 2013). In Australia, woody plant encroachment in Mitchell grasslands in Northern Territory and Queensland, for example, has created shifts in species composition (Burrows *et al.* 1986; Burrows *et al.* 1991; van Etten 2010). This is, according to van Langevelde *et al.,* (2003) is alternate vegetation state that can be irreversible (stable state) when left without intervention. Scott *et al.,*(2012) in his study of examining structure-composition of long unburnt sites within the northern Australian savannas following fire reintroduction for three years found swift reversibility to a structure of typical of frequent burnt savanna sites. Archer (1991) inferred from a space-for time substitution study of successional patterns and direct assessment using aerial photograph that in Parkland Texas, subtropical savannas had been converted into woodlands. In Brazil soil nutrients and fire-plant feedbacks influenced the forest-savanna coexistence (Silva *et al.* 2013). Silva *et al.,*(2010; 2013) found that nutrient supplies and adjustment to fire emphasize savanna and forest as alternate stable states where low fertility confines forest advancement over savannas. Meanwhile, the work of del Dantas *et al*., (2013) support the hypothesis that fire and plant feedbacks are regulating coexistence of forests and savannas as stable states. Their findings also highlight the important role of fire in regulating the savanna and forest ecosystems distribution in the tropics (de L. Dantas *et al.* 2013).

In summary, results from this study suggest that SDTF and savanna seems to be alternative ecosystem states with long unburnt savanna seeming to shift towards dry forest in terms of floristics. Further studies are needed to confirm or to obtain more evidence of the ecosystem transition identified here. Knowledge of this ecological shifting is important as both savanna and SDTF of Baluran National Park have significant roles in supporting the diversity of plants and big mammals including herbivores such as the Javan banteng (*Bos javanicus* subsp. *javanicus*) which is categorized as an endangered species on the IUCN Red List (IUCN 2008). Therefore conservation of endemic wildlife would be greatly benefited by their successful habitat management. Additionally, more research is needed in order to be able to delimit SDTFs properly, understanding their resilience to climate change (Tng and Standish 2017), and their response to repeated fires as well as other disturbances and threats.

Chapter 5: Spatial Patterns of Invasion by *Acacia nilotica* (L.) Willd. ex Delile in Baluran National Park East Java and its Future Projection across Eastern Indonesia

ABSTRACT

Acacia nilotica (L.) Willd. ex Delile. is reported as a noxious invasive alien species in Baluran National Park, East Java. This study examines patterns of invasion of *Acacia nilotica* in Baluran National Park over fourteen years using remote sensing. In addition, this study also develops a species distribution model (SDM) of *Acacia nilotica* based upon its naturalized distribution, to project the potential distribution of *A. nilotica* throughout the tropical environment of eastern Indonesia under current climate conditions. The research uses supervised classification of Landsat image as well as Geo-eye image analysis to describe and quantify spatial patterns of *A. nilotica* invasion in Baluran National Park, and a Generalized Linear Model to predict the species distribution to 2045 in eastern Indonesia. Results show that *A. nilotica* is rapidly advancing through the savanna of Baluran National Park, with the size of savanna size decreasing by ~1,361 ha over a 14 year period; meanwhile the *A. nilotica* stand has increased in size by ~1,886 ha over the same period and fire and grazing seem to play an important role in the invasion. Results also show that global climate change is likely to increase the potential distribution of *A. nilotica* in Indonesia and increases the area at risk of invasion. By year 2045, *Acacia nilotica* is most likely to spread to the eastern parts of Indonesia. In general, the model performance was good (AUC = 0.82). This climate model however, like many other SDMs, does not take into account biotic interactions as well as other environmental factors. Nonetheless climatic suitability is an essential requirement for successful establishment of an invasive species and species distribution models can disclose general patterns and convey useful estimates.

Keywords: Invasive alien species, *Acacia nilotica* (L.) Willd. ex Delile., remote sensing, species distribution model, climate change, Baluran National Park, tropical eastern Indonesia.

5.1. INTRODUCTION

Acacia s.l. (Mill.) is the prominent genus in the Fabaceae (Mimosoideae) with roughly 1,300 species distributed mostly in tropical and subtropical regions (Abari *et al.* 2012). One of common woody *Acacia* species of savanna ecosystems is *Acacia nilotica*. Although *A. nilotica* was the original type species for the genus *Acacia*, under a new classification system, the subgenus *Acacia*, to which *A. nilotica* belongs, has been renamed as the genus *Vachellia* Wright & Arnold (Dhileepan 2009; Kyalangalilwa *et al.* 2013). There is however controversy over the name change of species such as *Acacia nilotica* to *Vachellia nilotica* (Maslin 2008; Thiele *et al.* 2011). Although the re-typification has commenced, not everyone has adopted this classification (Maslin 2008; McNeill *et al.* 2007). Therefore, as the debate on nomenclature is not fully resolved and because *Acacia nilotica* is the name that many local people and land managers at Baluran use and are familiar with, so, in this thesis, I will continue using the name *Acacia nilotica* (L.) Willd. ex Delile. In its native range, *A. nilotica* is a widespread tree in the northern savanna regions of Africa and western Asia extending from Mali to Sudan and Egypt to Arabia and India. In *c.*1850 *Acacia nilotica* was first introduced to Indonesia in Java, being grown around teak plantations as a fire break. The species has been introduced to other parts of Indonesia such as Timor and Papua. *Acacia nilotica* is reported as dominant colonizer at Baluran National Park in East Java Province and Wasur National Park in Papua (Tjitrosoedirdjo 2008).

Although *A. nilotica* is known to locally dominate in African savanna (Brenan 1983), it has been scantily studied. One exception is the study by Skowno (1999) in the Hluhluwe Game Reserve, South Africa, who found that woody plant biomass has rapidly increased in Hluhluwe Game Reserve over the last 40 years and that open *A. nilotica* savanna is being replaced by broadleaf species, especially *Euclea* spp. In Australia, where this species has been introduced, it is widespread and locally dominant (especially along watercourses) in Queensland and Northern Territory, where it is a declared weed, with smaller outbreaks found in Western Australia, New South Wales and South Australia (DLRM 2015; Reynolds and Carter 1990). Radford *et al.* (2001) demonstrated the negative impacts of *A. nilotica* on savanna in northern Australia by excluding native flora and fauna and dramatically changing vegetation structure. *A. nilotica* can be threatening to savanna as its adult trees are apparently fire tolerant and can form thorny thicket formations (Burrows *et al.* 1991). This shift from open savanna to woodland and thicket is likely to result in environmental impacts such as reduction or loss of herbaceous plant communities, change in faunal habitat and altered hydrology (Radford *et al.* 2001a). Although adult trees of *A. nilotica* are apparently fire tolerant, the effects of fire on juvenile plants of this species are unknown. In addition, fire is often seen as a useful tool for the arrest of *Acacia* spp. and woody thicket formation and also for the maintenance of open savannas in many parts of Africa, however the effect of fire on *A. nilotica* are unclear and sometimes contradictory (Radford *et al.* 2001a).

Woody plant encroachment in the Mitchell grasslands of Northern Territory and Queensland has created situations where fire re-introduction may no longer be possible, thus causing shift in vegetation structure and composition (Burrows *et al.* 1986; Burrows *et al.* 1991; van Etten 2010). This is, according to Van Langevelde *et al.* (2016), an alternate vegetation state that can be irreversible (stable state). Savanna in Baluran National Park (BNP) has also been introduced with the *A. nilotica* in the late 1960s, where its original purpose was to create fire breaks to prevent fire to spread from the Baluran savanna to adjacent teak forest. However since then, *A. nilotica* has spread rapidly and now threatens the existence of the Baluran savanna with some areas having changed from open savanna to an almost closed canopy of *A. nilotica* (Figure 1) (Barata 2000; Djufri 2004). Over dominance of the woody species *A. nilotica* could shift the savanna into another ecosystem state, i.e. secondary/dry forest. This condition could put the large grazing mammals of Baluran savanna, such as water buffalo (*Bubalus* sp.), barking deer (*Muntiacus muntjak*), sambar deer (*Cervus unicolor*) and Javan banteng (*Bos javanicus* subsp. *javanicus*), at risk due to the loss of browsing and grazing fields (Sabarno 2002). The savanna within Baluran National Park has a significant role in supporting a diversity of plants and large herbivores such as the Javan banteng, a species of wild cattle found in Java and Bali which is categorized as an endangered species by the IUCN (IUCN 2008; Sabarno 2002; Suhadi 2009; Widayanti 2010).

Figure 1. Stand of *Acacia nilotica* in Bekol Savanna Baluran National Park, Indonesia

Many studies haved used remote sensing and geographical information systems (RS/GIS) to monitor condition and vegetation changes in savanna ecosystems (Chacón-Moreno 2004; Hudak and Brockett 2004; Sano *et al.* 2010; Stroppiana *et al.* 2003), however, not many have used RS/GIS to map invasive alien plant species in savanna-dominated landscapes. Setiabudi *et al*. (2013) conducted a spatial analysis of *A. nilotica* invasion in Baluran National Park, but they did not use remote sensing and there was no quantitative information on *A. nilotica* cover changes. Caesariantika *et al.* (2011) and Djufri (2012) conducted field studies of *A. nilotica* in Baluran National Park, which partly took place in the *Bekol* savanna (being a landmark of BNP where a vast area of savanna occurs and is named after the characteristic species 'Bekol', *Ziziphus mauritiana*); however these studies also ignored the capabilities of remote sensing. Siswoyo (2014) used Landsat imagery in his study in Baluran National Park, however, only one specific year was analysed and consequently there was no information of how much *A. nilotica* cover has changed.

Global climate change models (GCCMs) predict that by the end of 21st century, in comparison to averages prior to the 1980s, global warming will lead to: a 3-4 ^{o}C increase in mean temperature; a change in rainfall in many areas; greater rainfall variability; and significant changes in seasonality as well as severe weather events (IPCC 2007; Pachauri *et al.* 2014). At the regional scale, most species and ecological communities exist within a definable bioclimatic envelope, as at this scale spatial distributions are largely controlled by a set of climatic parameters, particularly those relating to precipitation and temperature (Molloy *et al.* 2013). When there are changes in these variables, the habitat value for that area will also change. The task of understanding how species and communities respond to changes is crucial. Climate change may facilitate alien species invasion into new areas, particularly for species from hot to warm climatic ranges introduced into areas currently marginal for that species in terms of climate (Sheppard *et al.* 2014).

Species distribution models or SDMs have the capability to assess current distribution and simulate climate-induced range shifts under different global change scenarios at the single-species and community levels (Crego *et al.* 2014). Therefore such models could identify areas at risk of further invasion by invasive alien species so that early preventative actions can be undertaken in a suitable approach. Franklin *et al*. (2014) describe steps in species distribution modelling as follows. Species occurrence data (such as presence-only, presence-absence or abundance data) are the response variable and environmental variables are the predictors used in a multiple regression-like modelling framework. Model can be fit into data space using a wide variety of statistical learning methods. Estimated parameters are then applied back to environmental data layers (mapped grids) to predict probability of species occurrences in geographical space (Franklin *et al.* 2014).

Species distribution models enable scientists and managers to predict future landscapes. Species distribution models can be applied in various fields such as in: 1) aiding invasive species prediction and management (Webber *et al.* 2011); 2) improving success in the conservation and reintroduction of endangered species (Adhikari *et al.* 2012; Molloy *et al.* 2016): 3) enabling adaptive management of protected areas (Mairota *et al.* 2014); and 4) restoring landscape connectivity (Gurrutxaga and Saura 2014). Predicting changes in species distribution with climate change has become a popular use of SDMs. For example, Molloy *et al.* (2013) constructed SDMs using bioclimatic variables to determine the impacts of a changing climate on the western ringtail possum (*Pseudocheirus occidentals*; Pseudocheiridae) in South-west Australia. Kritikos *et al.*(2003) conducted SDM using CLIMEX and predicted the potential distribution under future climate change scenarios for the invasive alien species *Acacia nilotica* subspecies *indica* in Australia. Their results showed that the potential distribution of the species in

Australia under current climatic conditions is vast and far greater than the current distribution. *Acacia nilotica* is a highly problematic species in Baluran National Park, and it is important to predict its distribution elsewhere in Indonesia under current conditions and under global climate change projections. Understanding the likely potential distribution of this obnoxious plant under current and future climate scenarios will enable policy makers and land managers to prepare appropriate strategies to manage the invasion.

Given the framework presented, the objectives of this chapter were to:

1. Map and quantify spatial patterns of invasion of *Acacia nilotica* locally in the Bekol Savanna and regionally in Baluran National Park using remote sensing imagery and examine whether the pattern has a similar match to the pattern of fire.
2. Predict the distribution of *Acacia nilotica* (L.) Willd. ex Delile. based upon its naturalized distribution in and outside of Indonesia and project the potential distribution of *A. nilotica* in eastern Indonesia under current climate conditions.

5.2. METHOD

5.2.1. Study Site

Baluran is located at the northern-most tip of East Java, on plains in the rain shadow of mountain ranges. In 1937 the Dutch Government instated this area as wildlife reserve (decree GB. No. 9 dated 25 September 1937 Stbl. 1937 No. 544) to conserve large mammals, mainly Javan banteng (*Bos javanicus* subsp*. javanicus*), that were inhabiting the surrounding areas. This decree was then reinstated by the Indonesian Agriculture Minister in 1962 (decree No. SK/II/1962 dated 11 May 1962) and then proclaimed as a national park in 1980. Baluran National Park covers a vast area of 25,000 ha and is located in Situbondo District, East Java Province. To its north, it is bordered by the Madura Strait and on its east side is bordered by the Bali Strait.

Baluran has a relatively dry climate and mainly consists of savanna (~40% by area), as well as dry forests and mangrove forests. According to the Schmidt and Ferguson classification, Baluran National Park has type F dry climate with temperature ranging 27.2-30.9°C, average relative humidity of 77%, and average wind speed of 7 knots. The rainy season generally runs from November to April, and the dry season is from April to October. Highest precipitation is usually in December to January. The Baluran savanna has volcanic soil types. In the dry season or during drought, the soil will crack (which suggest clays or loamy clays) with the depths of the crack reaching up to 80 cm.

5.2.2. Spatial Patterns of Invasion and Fire

A). Regional-Scale Patterns of Invasion and Fire in Baluran National Park

Satellite images for Baluran National Park (year 2000 and year 2014) were downloaded from Landsat 8 and Landsat 7 (http://earthexplorer.usgs.gov/) path 117, row 065. When selecting images to be download, I looked for dry season images which were not covered by clouds and tried to minimize the cloud cover percentage as much as possible with image quality level 9 (no errors detected, perfect scene). Most of the Baluran National Park areas had cloud free cover. Details of each image downloaded are outlined in Table 1.

LANDSAT images were georectified to reduce geometric distortion. I then chose band 6, 5, and 3 (for Landsat 8) and 5, 4 and 2 (for Landsat 7) and joined them into one image and then did merging of band 8 panchromatic to enhance and sharpen the composite image (pan sharpening). After layer stacking, cropping was done so that only

Baluran National Park area was shown. This result was then saved as a RGB image and used as the basis for classification (Figure 2).

Table 1. Details of images downloaded for remote sensing analysis.

Images	Source	Date acquired	Spatial Resolution	Image quality	Path/Row	Cloud cover	Total band	Image band stacked	Sensitivity
1	LANDSAT 7	09/7/ 2000	30 meter	9	117/065	25.83%	11	2, 4, 5	8 bit/pixel
2	LANDSAT 8	12/10/ 2014	30 meter	9	117/065	19.43%	11	3, 5, 6	12 bit/pixel

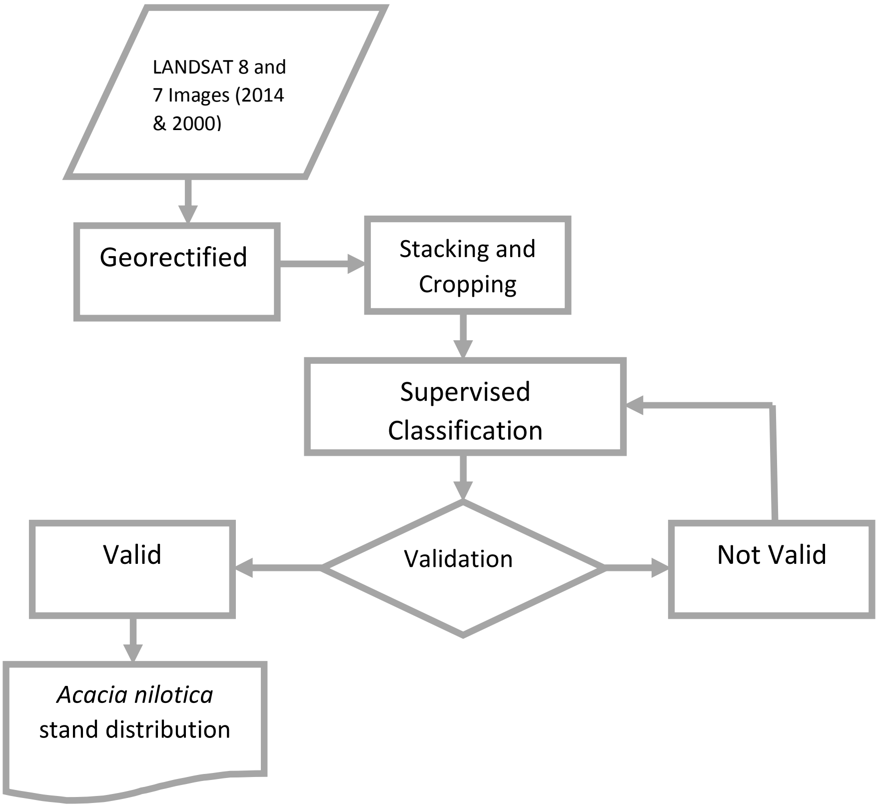


Figure 2. Flowchart showing steps in spatial analysis of *Acacia nilotica* in Baluran NP. Modified from Siswoyo, 2014.

Classification of the three vegetation types (savanna, forest and *A. nilotica* stand) was done using a supervised classification and a maximum likelihood approach within ENVI 4.5. Validation of the classification result was done by comparing it with ground survey points and also with Google Earth. Once classification finished, each class was converted to individual layer in a shape file to be analysed in Arc GIS 10.1. The data layers were projected into WGS 1984-UTM, Zone_49S. The size of each area/class was then calculated in Arc GIS for each image (year 2000 and 2014) to obtain information on whether or not there had been an increase in *A. nilotica* cover in Baluran National Park or more specifically to estimate the change in area covered by *A. nilotica* stands. Ground survey (for validation and check the maps) was done in October 2014 with a total 60 monitoring points laid randomly in three distinct vegetation types namely savanna, forest, and *A. nilotica* stand. Presence-absence of *A. nilotica* was recorded and geographical position was marked at the central of each plot with a handheld GPS.

MODIS burned area product (2000-2013) was used to describe regional-scale fire patterns in Baluran National Park. MODIS fire data is described in more details in Chapter 3. MODIS product are freely available from the web (NASA: http://modis.gsfc.nasa.gov/) with polygons exported to Arc GIS for spatial analyses. The MODIS data was then overlaid with the Baluran National Park land use map and ground truthing was conducted. I then extracted information on which locations were burned in the range period of 2000-2013 in Baluran National Park, the size of the burned areas, and when is the average peak time of the year when the fire occurred.

B). Local-Scale Pattern of Invasion and Fire in Bekol Savanna

Satellite images of the Bekol savanna for the period of 2000 to 2016 were obtained online using the Landsat Look Viewer (https://landsatlook.usgs.gov/) in spatially referenced Geo.tiff files. The identification of *A. nilotica* stand was based on prior knowledge of the plant's distribution from the thorough field observations in 2014. Expansion of *A. nilotica* stand was then calculated. To describe fire patterns in Bekol Savanna, instead of using coarse MODIS fire data, I used similar images from

Landsat look viewer which was used for the identification of *A. nilotica* stand. MODIS fire resolution is too large to use for Bekol Savanna, therefore I may not reliably infer the recent fire history of the savanna. A recent fire event was interpreted from the images by what was visually identified by darker colour shades in Bekol Savanna or its surroundings, as well as the shape of this dark area. The interpretation result was also checked through ground-truthing in the field.

5.2.3. Species Distribution Modelling of *Acacia nilotica*

I used the Biodiversity and Climate Change Virtual Laboratory (BCCVL; http://www.bccvl.org.au/) to model the distribution of *A. nilotica* in Indonesia based on climatic variables (Hallgren *et al.* 2016). The Global Biodiversity Information Facility (GBIF; http://www.gbif.org/) was used to obtain the species occurrence data for *Acacia nilotica* (L.) Willd. ex Delile (GBIF 2016). Although there are various subspecies of *Acacia nilotica* recognised and listed in the GBIF, I chose to use the *Acacia nilotica s.l.* (L.) Willd. ex Delile. as this is what is recorded to be present at Baluran National Park in East Java, Indonesia (Tjitrosoedirdjo 2008) and there is uncertainty over the various subspecies present across Indonesia. GBIF is an international open data infrastructure funded by various governments. The database has 879 occurrence records of *Acacia nilotica* (L.) Willd. ex Delile, with 232 of these occurrences being geo-referenced. This data then imported to Biodiversity and Climate Change Virtual Laboratory (BCCVL) with sites of dubious location removed (sites substantially outside the species known range, eg Europe, or non-terrestial). This species presence data acts as the response variable.

The predictors are various available environmental variables. In this simulation I used Worldclim current conditions (1950-2000) at 2.5 arcmin resolution. I started by selecting all environmental variables available in the BCCVL and then eliminated those which were not influential as measured by response curve. This process narrowed down the climatic variables to only seven, namely: maximum temperature of warmest month (B05), minimum temperature of coldest month (B06), mean temperature of wettest quarter (B08), mean temperature of driest quarter (B09), precipitation of wettest month (B13), precipitation of driest month (B14), precipitation seasonality (B15) and temperature seasonality (B04). Mean conditions at their most extreme time of the year tend to make better variables for SDMs as they are generally more influential in determining the limits of occurrence of species (Crego *et al.* 2014). Precipitation and temperature variables represent important factors determining vegetation range and abundance (Krebs 1985; van Steenis 1972), and *A. nilotica* distribution in Australia correspond well with these climate variables (Kriticos *et al.* 2003). The Worldclim collections consists of a set of global climate layers (climate grids) covering all global land areas except Antarctica. They are in the latitude/longitude coordinate reference system (not projected) and the datum is WGS84. The data layers were generated through interpolation of average monthly climate data from weather stations producing high quality data and there are two main components to the Worldclim collection: current climate and future climate (Hijmans *et al.* 2005).

The Species Distribution Model (SDM) allows us to investigate the potential distribution of a species under current climatic conditions (Gray *et al.* 2010; Hallgren *et al.* 2016). The BCCVL currently provides 17 different algorithms across 4 different categories to run species distribution model (e.g. profile model, statistical regression model, machine learning model and geographic model) (Gray *et al.* 2010; Huijbers *et al.* 2016; Richmond and Huijbers 2016). True absence data is not available for the species of interest; therefore I used random pseudo-absence data option available in BCCVL's

SDM feature (Huijbers *et al.* 2016). In this study I used statistical regression model, namely Generalized Linear Model (GLM), to process SDM because it accommodates regression model for data with non-normal distributions, fitted with maximum likelihood estimation. This model produces estimates of the effect of different environmental variables on the distribution of a species. The model uses all the data available to estimate the parameters of the environmental variables, and construct a function that best describes the effect of these predictors on species occurrence (Huijbers *et al.* 2016). The suitability of a particular model is often defined by specific model assumptions. The prediction is visualised as the suitability of a grid cell on a scale from 0 to 1, where 0 refers to very low suitability and 1 refers to very high suitability (Huijbers *et al.* 2016). Results from BCCVL are .tiff files which then processed further using Arc GIS 10.1. Raster values created and classified, then given different colours to clearly differentiate areas with different habitat suitability index.

The primary output of a SDM is a map that shows the predicted distribution of *A. nilotica* under baseline conditions. It is important to note that this not really a prediction of where the species occurs, but rather the distribution of suitable habitat as defined by the environmental variables (in this case mean current climate conditions) included in the model. A secondary output is the response curve. Response curves are plots that show the relationship between the probability of occurrence for a species and each of the environmental variables (Huijbers *et al.* 2016). For each plot, the response is modelled for one environmental variable while the other environmental variables are held constant at their mean. The x-axis represents the range of values of the environmental variable, and the y-axis gives the probability of occurrence on a scale from 0 (low probability) to 1 (high probability) (Richmond and Huijbers 2016). Model robustness was evaluated using the AUC (Area Under the Curve) of the ROC curve

(Receiver-Operating Characteristics), which is a nonparametric threshold-independent measure of accuracy commonly used to evaluate SDMs (Bertelsmeier and Courchamp 2014). The ROC plot is a graph with the False Positive Rate (1-Specificity) on the x-axis and the True Positive Rate (Sensitivity) on the y-axis plotted across the range of threshold probability values. The closer the ROC curve follows the y-axis, the larger the area under the curve, and thus the more accurate the model. The value for ROC is the area under the curve (AUC), and is calculated by summing the area under the ROC curve. A value of 0.5 represents a random prediction, and thus values above 0.5 indicate predictions better than random (Huijbers *et al.* 2016). I interpreted the AUC score as follow: values above 0.9 is excellent, good 0.9 > AUC > 0.8, fair 0.8 > AUC > 0.7, poor 0.7 > AUC > 0.6 and fail 0.6 > AUC > 0.5 (Crego *et al.* 2014; Sweets 1988).

A second SDM was then conducted to predict the distribution of *A. nilotica* under potential future climatic conditions. In BCCVL this is called the "Climate Change Experiment"and leads to predictions of where *A. nilotica* could occur in the future under a particular climate change scenario. This analysis uses the results from the first SDM (including the same climatic variables), and projects that distribution for a certain year in the future with the climate information from one of several climate models. In this study, I used the RCP 8.5 (business as usual) greenhouse gas emissions scenarios to influence the climate model using the CSIRO Mark 3.0 model with a 30'' (~1km) resolution. I projected the SDM to year 2045.

5.3. RESULTS

5.3.1. Spatial Patterns of Invasion and Fire

A). Regional Scale Patterns

Savanna was clearly shown in the Landsat satellite image (Figure 3) as mix of bright glossy white and light – semi dark purple-ish colours, while *A. nilotica* stand was seen as mix of semi-dark green and light to dark brown. Overall accuracy of the classification was 80%. In the year 2000, *A. nilotica* mainly occurred around the Bekol Savanna area and surrounds (Figure 4). Fourteen years later *A. nilotica* has spread far north and also south of the National Park areas, invading not only savannas but also dry forests (Figure 4). Over fourteen years, the area of savanna size has decreased (Figure 5), from approximately 6,510 ha in 2000 to 5,149 ha in 2014 (- 1,361 ha). Conversely, the *A. nilotica* stand has increased (Figure 5) in size by around 1,886 ha (from 1,742 ha in 2000 to 3,628 ha in 2014). Landsat 8 image analysis for October 2014 shows that *A. nilotica* expands as a dense stand in the north-west and east of the national park (Figure 4). As of 2014, *A. nilotica* stands occupied an area of 3,628 ha or about 14.5% of the total area of Baluran National Park.

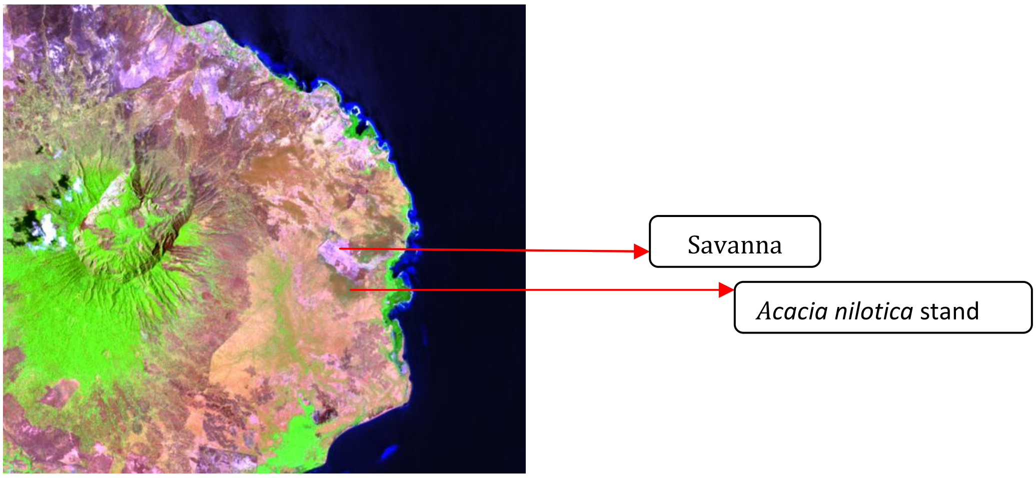
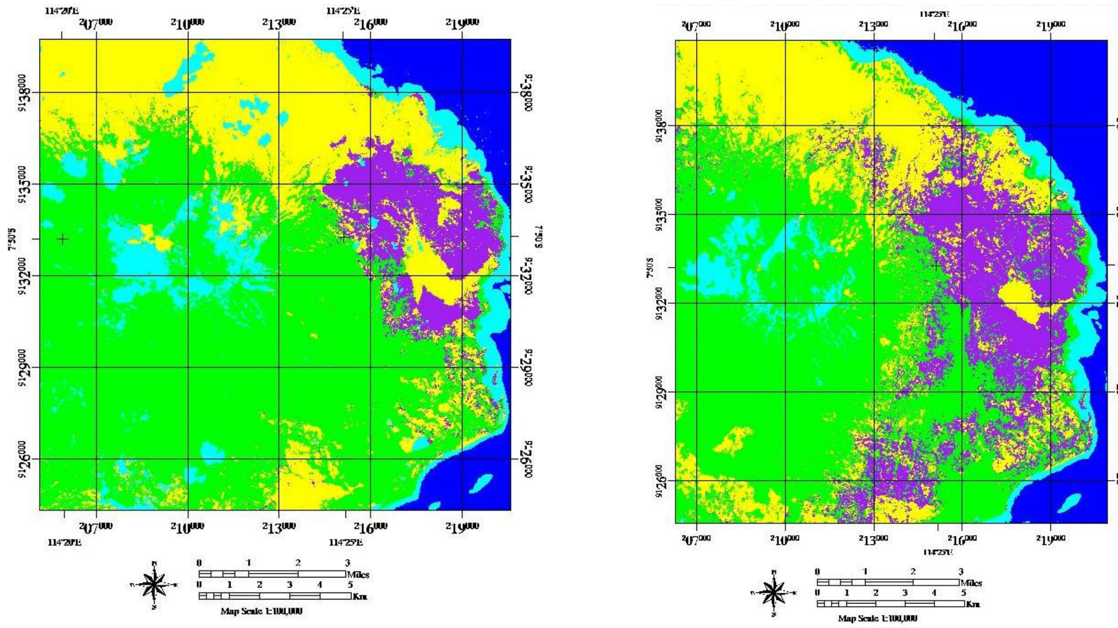


Figure 3. Different colour of savanna and distinct *Acacia nilotica* stand was shown in this combination of band 6, 5 and 3 of LANDSAT 8 image of Baluran National Park, 2014.



(A) (B)

Figure 4. Expansion of *Acacia nilotica* stands (purple colour) in Baluran National Park over 14 years using classified image. Left (A) *A. nilotica* distribution in year 2000, right (B) *A. nilotica* distribution in 2014. Yellow area in the middle is Bekol savanna area and green colour is forest. Light blue colour is cloud cover.

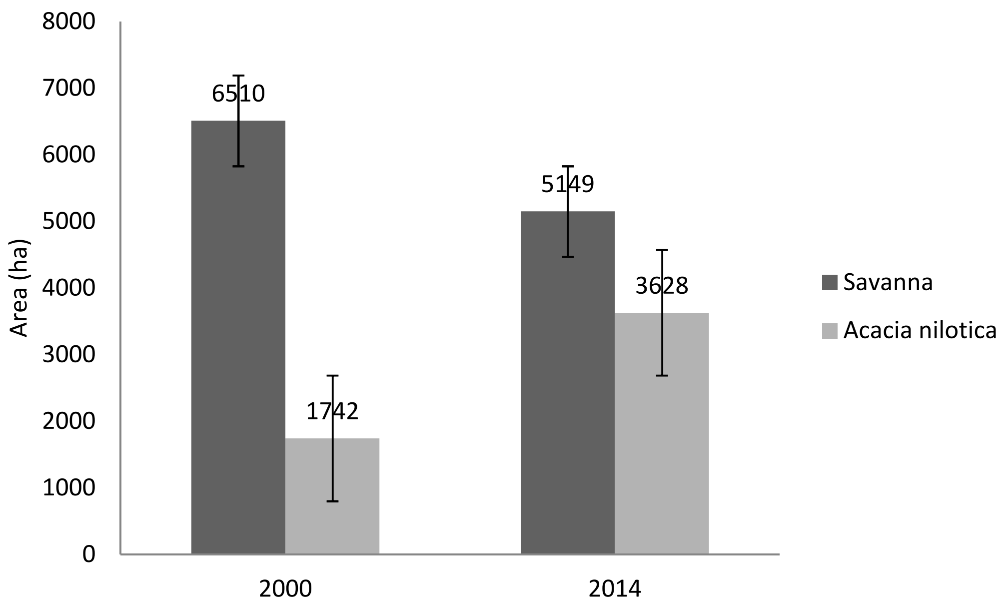


Figure 5. Changes in area (ha) (data extracted from the created map, see Figure 4) for savanna and *Acacia nilotica* stand between year 2000 and 2014 in Baluran National Park.

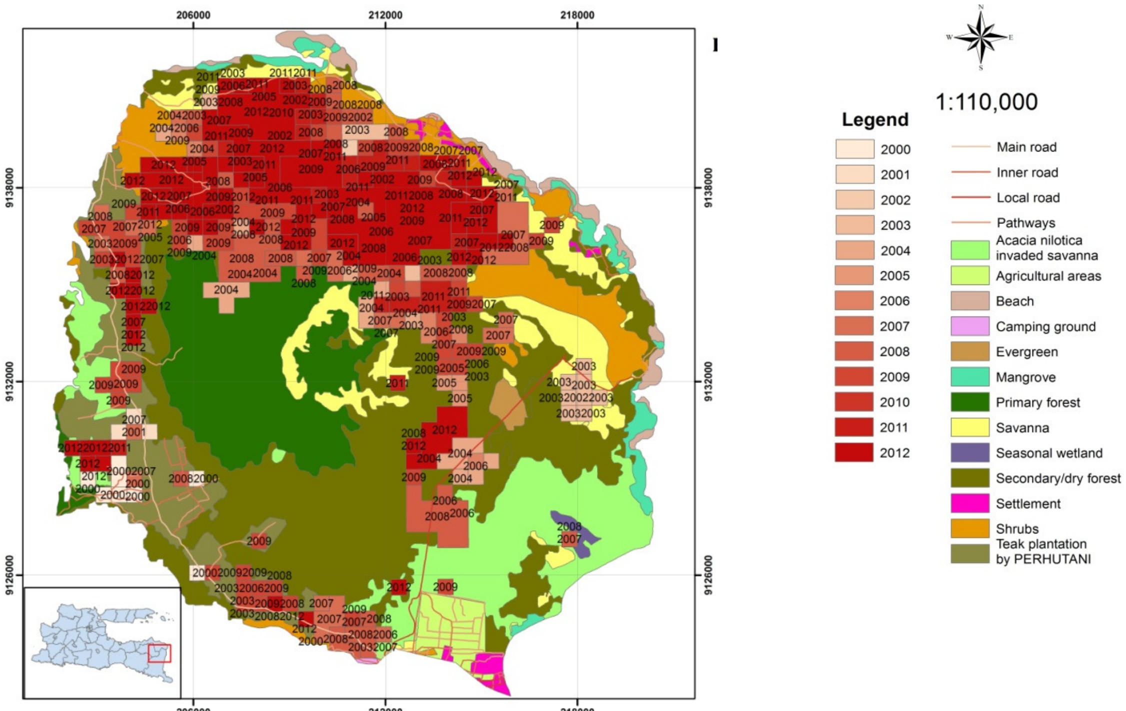


Figure 6. Spatial patterns of fire in Baluran National Park, East Java, Indonesia, 2000 – 2012 showing distribution of burned areas (as detected using MODIS burned area product) which are mostly concentrated in the northern part of the national park. Note the year of most recent fire is shown (many areas burnt more than once during the study period).

Fire spatial and temporal patterns in Baluran National Park from year 2000 to 2013 were shown in fire distribution map (Figure 6). Fires were mostly concentrated on the north side where vast areas of savanna were located. There were 1,711 fire 'hotspots' detected in Baluran. Fires detection were higher in dry months June, July, August and September, with August being the month which usually fires were detected the most. Seasonally dry tropical forest burned the most (2,629.6 ha) which was then followed by savanna (1,401.4 ha), primary forest (600.9 ha) and invaded savanna by *A. nilotica* (394.9 ha).

B). Local Scale Patterns

Spatial analysis of the satellite images showed distinct patterns of *A. nilotica* stand spread. *Acacia nilotica* stand could be readily distinguished from the other vegetation types (Figure 7). The map shows the dynamics of *A. nilotica* stand with an increasing trend of expansion both on the northside of the road as well as on the south of the road in Bekol Savanna. In 2002, the size of *A. nilotica* stand around the Bekol Savanna was 91.75 ha. This figure had doubled by 2015 (Figure 8). Across the period 2002 – 2015, savanna areas in Bekol decreased by 40%. Fires burnt vary amounts of the Bekol savanna throughout the period (Figure 7). Burned area size was highest in 2002 and was the lowest in 2015 (Figure 8). Fires in Bekol Savanna mostly took place at the peak and late dry season (August-September-October).

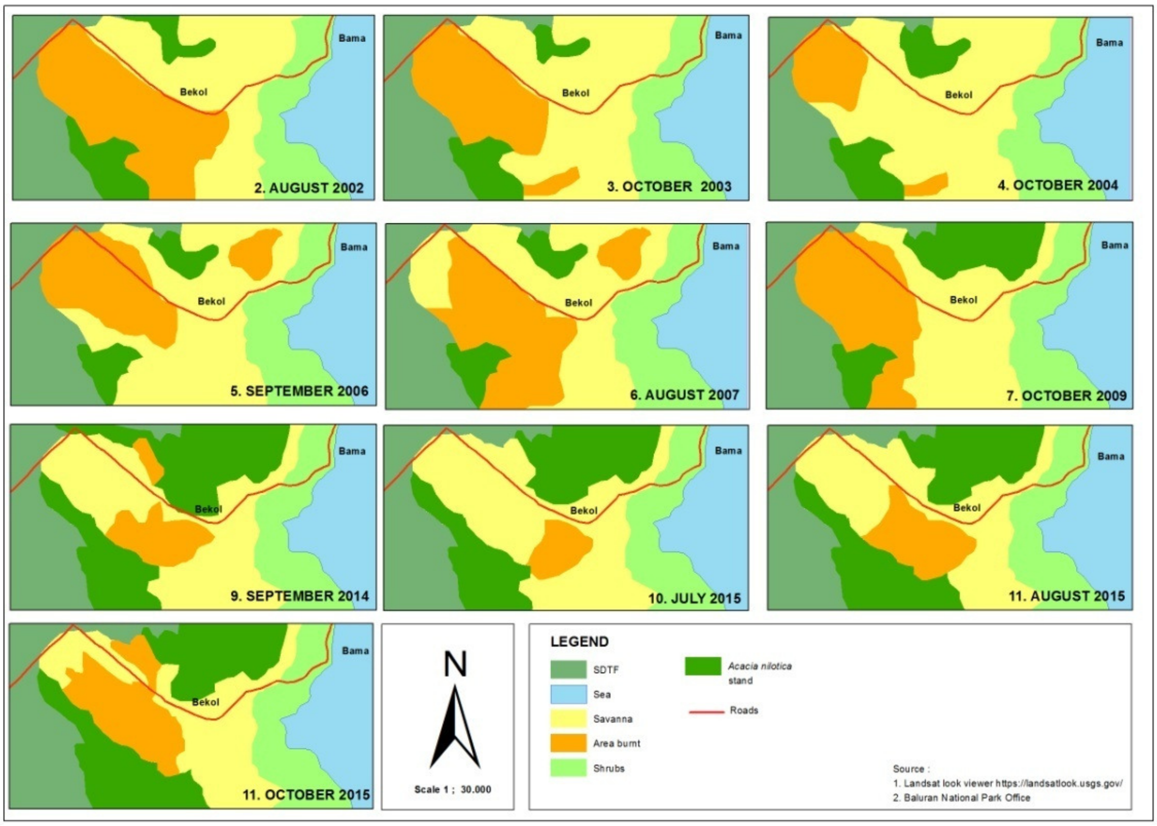


Figure 7. Map showing spatial and temporal patterns of *Acacia nilotica* invasion and fire in Bekol Savanna, Baluran National Park. SDTF in legend refers to Seasonally Dry Tropical Forest. Date of imagery is shown. Note that some years are missed because clear images for that certain years are not available.

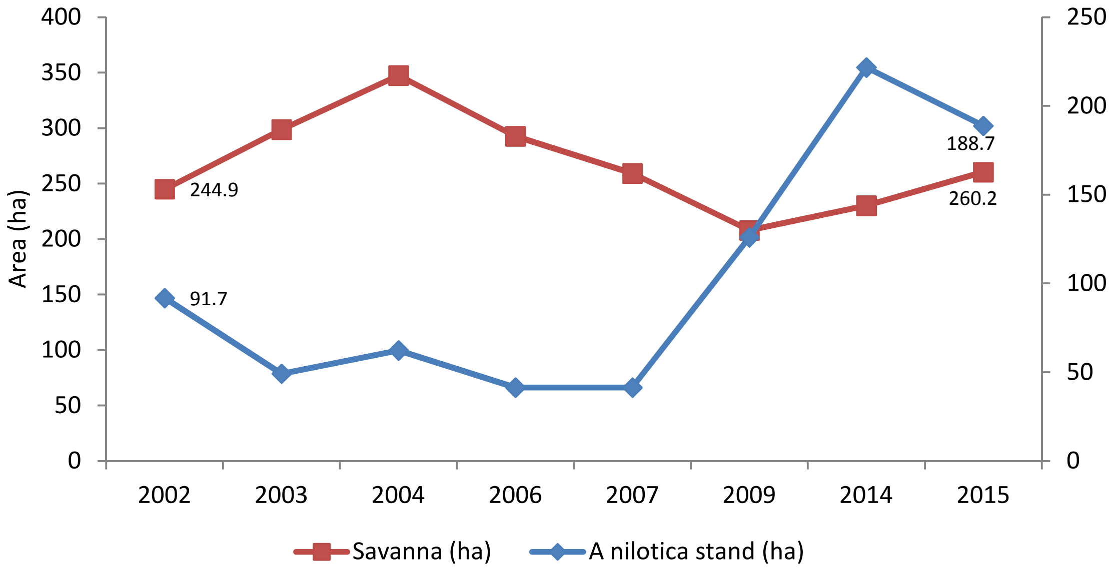


Figure 8. *Acacia nilotica* stand area size (ha) and savanna annual burnt area (ha) in Bekol, Baluran National Park.

5.3.2. Species Distribution Modelling of Acacia nilotica

The potential distribution of *A. nilotica* in Indonesia under current climatic conditions is vast and far greater than the current distribution. Under current climate, it is predicted that *A. nilotica* has potential to spread to eastern parts of Indonesia, although some parts of Java, especially along the south coast of East and Central Java, are also predicted to be suitable for *A. nilotica*, including the area of Baluran National Park (Figure 9). In eastern Indonesia, *A. nilotica* is predicted to be suitable to inhabit several sites in south-eastern Bali, south west of Lombok, south eastern Sumba, most of Kupang Province, the west side of Timor Island and also some parts along the south coast of west Papua. In West Papua, the species is predicted to be potentially suitable to inhabit (habitat suitability ~0.64) the southern coast to Merauke. It is also interesting to note that small islands just off the coast between Merauke and Tual are predicted by the model to be potentially suitable (~0.61) habitat for this IAS (Figure 9). There is a trend of increasing mean suitability index (0.47 to 0.6) as we move from central to eastern parts of the archipelago (i.e. from East Java, to Bali, Lombok, Sumba, Timor and Papua – see rectangle in Figure 9).

Global climate change is predicted to increase the potential distribution of *A. nilotica* in Indonesia. Bali, Lombok Island (West Nusa Tenggara NTB), Sumba Island and Kupang (East Nusa Tenggara), as well as West Papua, all have increased area at high risk of invasion. On Sumba Island (Figure10), the current prediction has one area (*Baing* District in the south eastern Sumba) where the HSI (Habitat Suitability Index) is in the range of 0.7 – 0.9, and by 2045 it is predicted that *Melolo* District in north eastern Sumba will also have similar HSI to the *Baing* District. For Kupang on the west Timor Island, it is detected that there will be an increase on the HSI around the Tenau and Oesau Districts by 2045 (Figure 10).

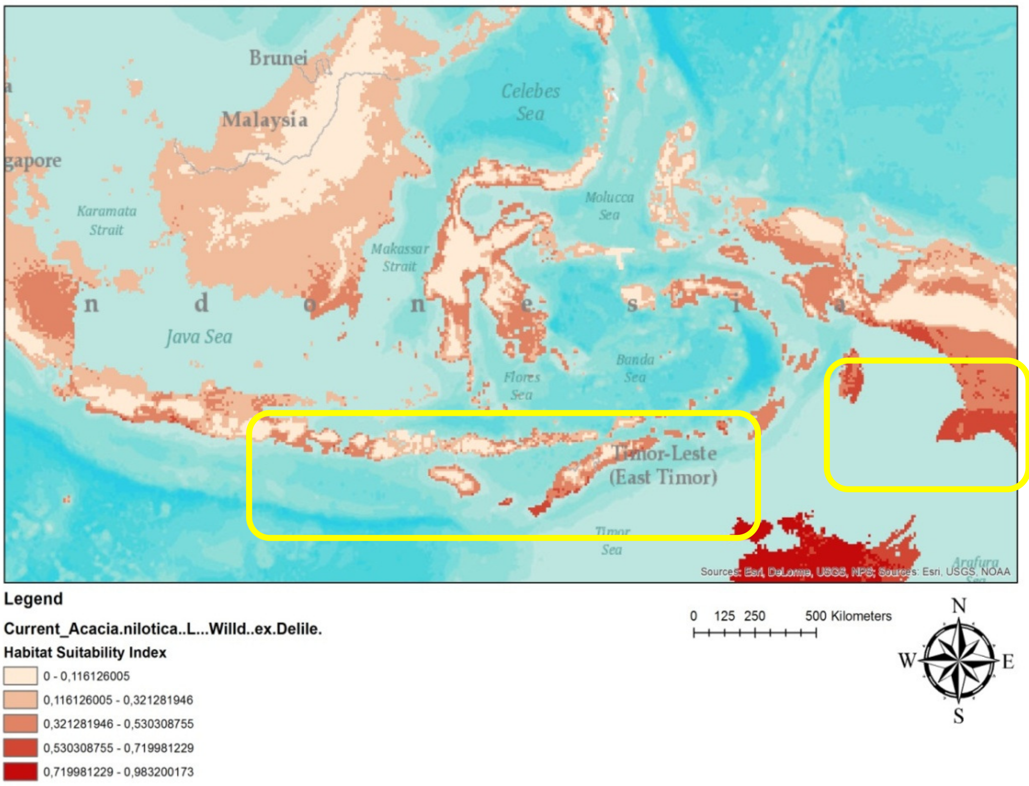
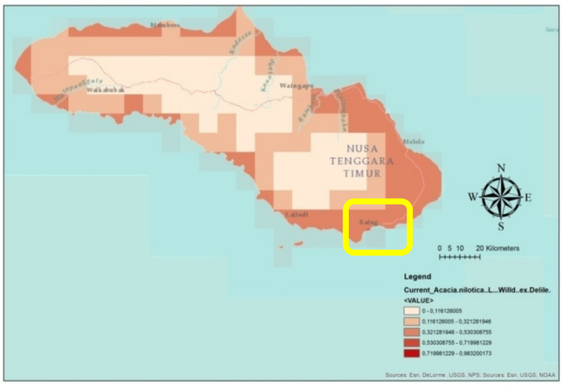
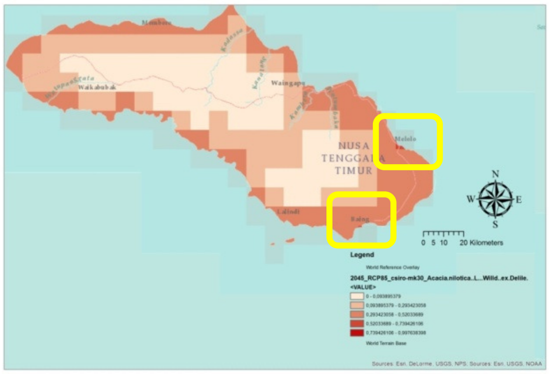


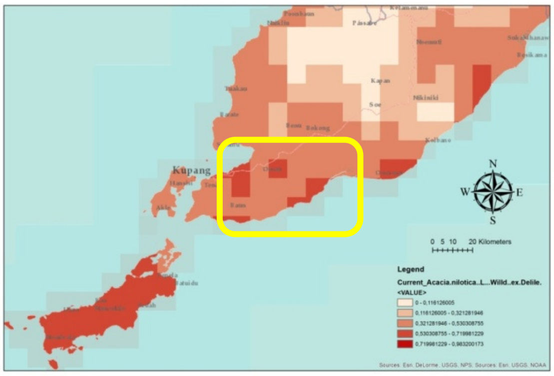
Figure 9. Predicted current distribution (habitat suitability map) of Invasive Alien Species (IAS) *Acacia nilotica* (L.) Willd. ex Delile.in Indonesia under current climate condition using Generalized Linear Model (GLM) algorithm in BCCVL. Darker areas representing a higher likelihood that the species can occur. Rectangular yellow lines point out features/regions highlighted in the text.

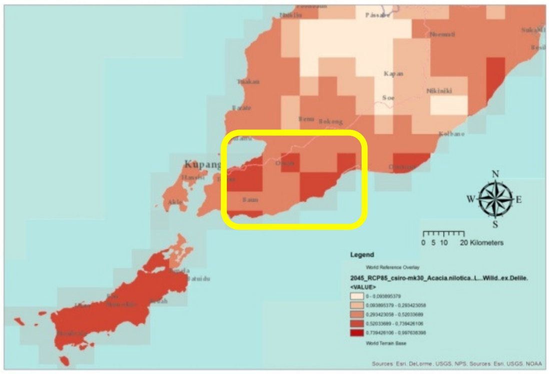




Sumba
Current

Sumba
2045





Kupang
Current

Kupang
2045

Figure 10. Projection of *Acacia nilotica* (L.) Willd. ex Delile. species distribution/Habitat Suitability Index (HSI) on Sumba Island and Kupang District in East Nusa Tenggara Province, Indonesia by 2045. Left: Predicted current distribution/current HSI. Right: Projected distribution/projected HSI in 2045. Yellow squares show areas where the HSI changed with greater areas at high risk of invasion. Darker areas represent a higher likelihood that the species can occur.

All of the chosen climate variables were found to be responsive to *A. nilotica* distribution (Figure 11). The response curves in this plot show that the probability of occurrence of *A. nilotica* follows a bell curve for each specific variable. *Acacia nilotica* occur in areas which pronounced seasonal climates. It can grow in places with low rainfall and high rainfall (10 – 150 mm/month). It does not like extreme cold or frost, it can grow in areas where minimum temperature of the coldest month is around 12 to 13 °C and it also grows in areas where the maximum temperature of the warmest month is around ~35°C. In general, my model performed well with AUC values (0.82) which is in the recommended range of 0.8 – 0.9 (Elith and Leathwick 2009); Figure 12).

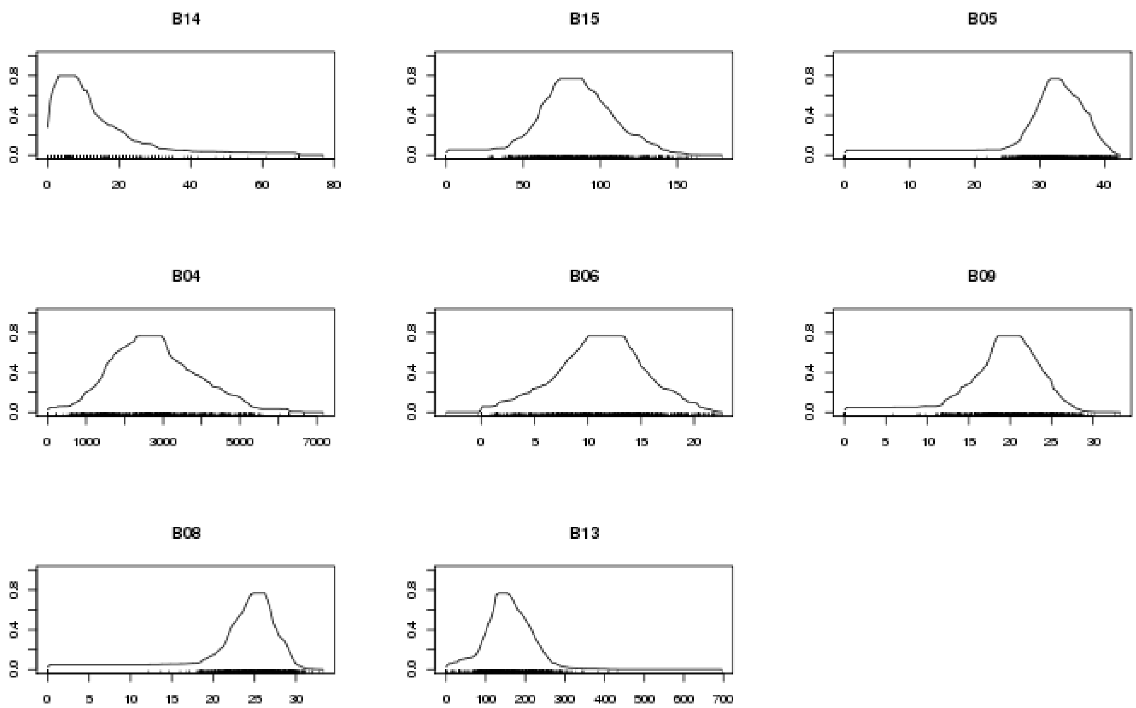


Figure 11. Response curves for *Acacia nilotica* (L.) Willd. ex Delile. distribution model. B04 to B14 in the figure refers to the variables. B04 = temperature seasonality, B05 =maximum temperature of warmest month, B06 = minimum temperature of coldest month, B08 = mean temperature of wettest quarter, B09 = mean temperature of driest quarter, B13 = precipitation of wettest month, B14 = precipitation of driest month, and B15 = precipitation seasonality.

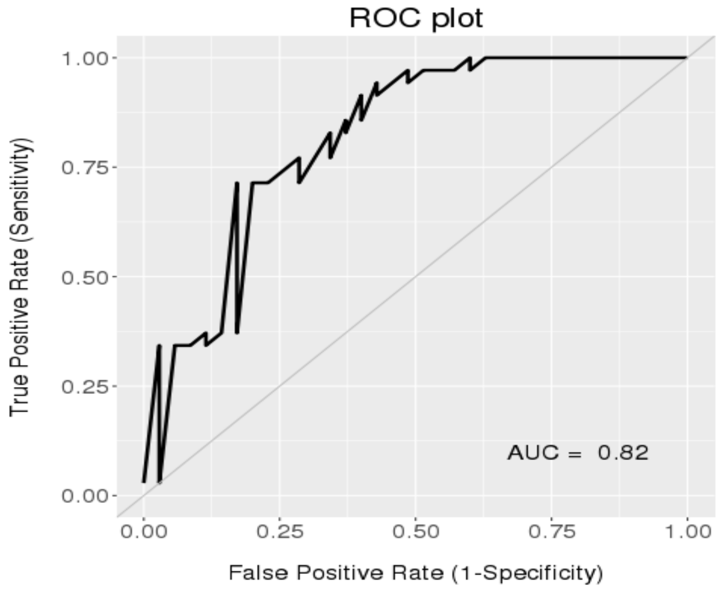


Figure 12. ROC plot which shows AUC value for *Acacia nilotica* (L.) Willd. ex Delile. model using GLM in BCCVL

5.4. DISCUSSION

This study investigating spatial patterns of invasion of *Acacia nilotica* in the savanna of Baluran National Park, East Java, demonstrated the following: i) spatially, *A. nilotica* is rapidly spreading into and transforming the savannas in Baluran National Park where it was observed over a fourteen year period to decrease the savanna area by 21%, and increase the area of *A. nilotica*-dominated stand by 52%; and ii) fire may play an important role in regulating the invasion. The study has also shown that global climate change is likely to increase slightly the potential distribution of *A. nilotica* in Indonesia, increasing the area at high risk of invasion. By year 2045, *A. nilotica* is most likely to spread to eastern parts of Indonesia.

Using remote sensing analysis, a supervised classification of Landsat images across the period 2000 to 2014 clearly showed the decrease in savanna areas and corresponding increase of *A. nilotica* stand throughout Baluran National Park. *A. nilotica* grows in a structurally homogeneous and mostly single-species stands in the north-west and eastern parts of the national park. In the year 2000, *A. nilotica* mainly occurred in a lowland area around the Bekol Savanna area and its surrounding. Fourteen years later *A. nilotica* has spread far north and also south of the national park areas, invading not only savannas but also dry forests. The predicted area of the *A. nilotica* spread in this study may be smaller than the actual occurrence in the field. This is firstly because Landsat image could not identify *A. nilotica* that grows individually or in a small clumps that may be smaller than the Landsat image pixel's size representation; secondly the imagery cannot detect where *A. nilotica* may be sparsely distributed – e.g. saplings spread across savanna. The resolution of the Landsat images is 30 m, which means that a vegetation type, such as the *A. nilotica* stand, needs to cover most of a 900 m^2 pixel for it to be detected in that pixel (Siswoyo, 2014). In other words, *A. nilotica* needs to be the dominant cover type in a pixel for it to be recognised as *A. nilotica* cover. The maps also show that vast areas of savanna in the north-east side of the national park have somehow 'escaped' the invasion. I noted during field observations that there are actually a few scattered young *A. nilotica* within the savanna on this north-east side, which potentially means this area may be colonised in the near future. At a more local scale, the rapid invasion of *A. nilotica* was measured in the Bekol Savanna. In this popular landmark savanna of the Baluran National Park, over just 13 years, there was a decrease in areas of the Bekol Savanna by 40%, whereas *A. nilotica* stand areas in Bekol expanded by 51%. These expansion areas at the boundary between savanna and *A. nilotica* stands, as explained in the previous chapter (Chapter 4), were mainly comprised of younger *A. nilotica* (mostly sapling sizes, but also seedlings), thus demonstrating an active invasion front (Fuentes-Ramirez *et al.* 2011). I have also shown in chapter 4, that *A. nilotica* is spreading into dry forest (seasonally dry tropical forest/SDTF) of Baluran National Park. Beside *A. nilotica*, this area was also mixed with other woody species, but which had less dominance, namely *Azadirachta indica, Ziziphus mauritiana, Thespesia lampas, Polytrias amaura* and *Dichanthium coricosum*. Up to date there have been few reports that noted the locations and patterns of spread of *A. nilotica* into savannas and forests (Djufri 2004; Sabarno 2002; Setiabudi *et al.* 2013; Siswoyo 2014).

We know now that *A. nilotica* is rapidly invading Baluran savanna, but, what are the external factors and what are characteristics of the species that encourage the spread and the domination of *A. nilotica* in the savanna? This question needs further research into community ecology and seed ecology of *A. nilotica*, which will be addressed in the next chapter. From this chapter results, however, it shows that there may be association between *A. nilotica* invasions and fire. Examination of fire history of the national park using MODIS showed that savannas on the north-east side and also at Bekol experienced frequent fires during the study period. The Baluran management office is trying as much as possible to maintain these savannas on the north-east and also on Bekol area by frequently practicing controlled burning (Caesariantika *et al.* 2011; Sabarno 2002). Regular fire seems to have halted the expansion of *A. nilotica* on the north-east side. However, fire perhaps has been less successful in halting the invasion into the Bekol savanna. Single burns may in fact promote initial *A. nilotica* spread by promoting germination, so regular fire is probably required to also prevent seedlings developing into saplings. However a rapid reduction in burning frequency and extent could facilitate *A. nilotica* spread as seedlings may make it to sapling stage where they can resist and/or resprout after fire. This may explain the spread of *A. nilotica* in the Bekol savanna from 2009 to 2014 as there was peak and then major reduction in burn area over this time frame. Besides fire, many other factors also perhaps play substantial role in the advance or progression of the invasion. Factors such as *A. nilotica* density and life stages may also affect the success of fire in halting the invasion. Fire may be effective tool to use to remove *A. nilotica* when the species is still at its seedlings or younger stage and less dense cover. Whereas in a savanna where mature *A. nilotica* with thick density and high cover, fire perhaps is not effective to eradicate this invasive species (and additionally the species is likely to resprout at this age). In this situation, fire may become an agent of facilitation which ultimately leads to further invasion.

The species distribution model (SDM) shows the current predicted distribution of *Acacia nilotica* across Indonesia. Areas with the highest concentration of potential invasion are mainly located along the coastline, especially in the south-eastern parts of Indonesia archipelago, such as East Java, Bali, Lombok, Sumba, Kupang and West Papua. This result agrees with current locations of *A. nilotica* recorded in Indonesia in the literature and also based on my encounters in the field. Fisher (2010) mentioned the presence of *A. nilotica* in savanna in Kupang, East Nusa Tenggara, Indonesia in 2010. Additionally, during my fieldtrip to Kupang in 2015, I also encountered this thorny *Acacia* in a savanna near a roadside to Tablolong beach. In his expedition in 2010, Arinasa (Pers. Comm.) noted the presence of *A. nilotica* in Bali Barat National Park. Seeds and fruit samples collected from Bali Barat National Park have also been processed and registered at the Hortus Botanicus Baliensis in Bali Botanical Garden. The species has spread also outside of Java Islands, to islands such as Timor and Papua. *Acacia nilotica* is reported as dominant colonizer at Baluran National Park in East Java Province and Wasur National Park in Papua (Tjitrosoedirdjo 2008).

Acacia nilotica is found at various temperatures, but is reported to be very sensitive to frost but will grow in areas where the mean monthly temperature of the coldest month is 16°C (FAO 2014). This is in line with my findings. My model shows that *A. nilotica* habitats are very seasonal; it follows an optimum curve for the selected temperature between ~13 and 35 degrees Celsius. Such climatic factors have some connection with the main land type where this species occurs (which is mostly in savanna or grassland) as places that have high seasonality tend to be savanna, as can be seen in Baluran National Park East Java and on Sumba Island (Bond and Keeley 2005; Sutomo *et al.* 2016). Savanna is the dominant type of landcover on Sumba Island and it is also the most burnt (Chapter 3). Sumba Island receives an average precipitation of 900 mm/year and experiences more than 5 months of dry period, being located in region A (southern monsoonal region) of Indonesian climate classification (Aldrian and Susanto 2003; Hanifah 2014). The high average temperatures and low rainfall with prolonged dry season, combined with local practices, means fires are more prominent in this landscape (Fisher *et al.* 2006; Monk *et al.* 2000). Most fires in Sumba occur in the eastern part of the island (Chapter 3) where the potential for *A. nilotica* was found to be the highest.

The current SDM model may have provided more insight and finer-scale predictions when overlaid with other environmental and landscape layers. Specifically, modelling could be improved by including variables such as soil type and topography, as well as other variables. *Acacia nilotica* is found in savannas and grasslands growing on various soil types (DAFF 2014; FAO 2014; GBIF 2016). In some places in Indonesia, *A. nilotica* is found in savanna ecosystems with volcanic - type soil, such as Baluran National Park (Sutomo *et al.* 2015), but it is also found in other savannas where the soils are not of volcanic origin, such as in Sumba and Kupang in NTT Province.

The model projection shows a fairly small increase in potential distribution of *A. nilotica* in south eastern parts of Indonesia by 2045 with climate change. Although the model had moderate to good performance (based on AUC), it shares a number of fundamental assumptions common to all species distribution models. These models do not take into account biotic interactions or microclimate conditions which may decide at a very local scale whether or not a particular invasive species eventually becomes established in a given area (Bertelsmeier and Courchamp 2014). Also important is a vector or means of spread (distribution of seeds and other propagules). For example, interaction between *A. nilotica* with herbivores has benefited *A. nilotica* (Chapter 6)*.* The lowland savanna area is an important habitat for big mammal grazers such as wild buffalo, deer and javan banteng. Invasion of *A. nilotica* in these areas may have caused changes in the feeding behaviour of these grazers and may have been causing the spread of this invasive species further in the national park. *Acacia nilotica* is also unpalatable to herbivores as this plant possesses thorny spikes on branches which make it difficult for the herbivores to consume the leaf. However, the pods that drop to the ground are usually consumed by herbivores during the prolonged dry period when fresh shoot and grasses are scarce (Sutomo *et al.* 2015; Sutomo *et al.* 2016; Tjitrosoedirdjo *et al.* 2013). At the end of the wet season toward the dry season, mature *A. nilotica* pods drop from the trees and are consumed by herbivores such as water buffalo that spread *A. nilotica* further in the national park (Sutomo *et al.* 2016; Tjitrosoedirdjo *et al.* 2013). Nonetheless climatic suitability is an essential requirement for successful establishment of an invasive species and species distribution models can disclose general patterns and convey useful estimates (Bertelsmeier and Courchamp 2014).

In addition, other environmental factors such as soil nutrients, land use changes and disturbance agent such as fire can improve model predictions. Moreover our understandings on the capability of *A. nilotica* to adapt to the changes in these factors are still limited (but see Kriticos *et al*., 2003). Based on their species distribution modelling and climate change projection for *A. nilotica* (subsp. *indica*) in Australia, it is expected that there may be increases in water-use efficiency of the species due to increased atmospheric CO_2 concentrations, which allowing it to invade more xeric sites further inland, and increased temperatures. *Acacia nilotica* is frost intolerant, grows in areas where the mean monthly temperature of the coldest month is 16° C and it can endure temperatures up to 50° C (Kriticos *et al.* 2003). Conservationists should not deter from using the predictive power of SDMs although modelling limitations applies. A dynamic SDM, based on well-surveyed populations responding to changes in known-critical parameters, is one of the best existing tools for conservationists to visually suggest future conditions. Users should be aware of limitations, but the model insights are a vital preliminary point for decision making (Carvalho *et al.* 2011).

Finally, this research has demonstrated the usefulness of remote sensing technology to establish the patterns of spread and current distribution of *A. nilotica* cover. This is one of the advantages of remote sensing and GIS, where the conventional approach of vegetation analysis is difficult and expensive to conduct directly in the field (van Etten and Fox 2004; van Etten 1998). Remote sensing and GIS have been also used in Indonesia and a range of literature has demonstrated their use for various ecological studies such as to assess habitat suitability (Gamasari 2007), land use and cover change (Lavigne and Gunnell 2006), species conservation (Iskandar *et al.* 2012), and many others. In terms of savanna ecosystem studies, remote sensing could provide insight into the patterns of habitat fragmentation due to invasive alien species (IAS), as well as fires. This study is the first to model and highlight regions with a high risk of invasion by invasive alien species *Acacia nilotica* (L.) Willd. ex Delile in tropical environment of southeast Asia, specifically the central and eastern parts of Indonesia. Future research should aim to develop extrapolative modelling based on traits related to invasiveness that can be combined with the current model of predicting establishment likelihood.

More research is needed on how to use SDMs in the service of informing public policy, scenario analysis with stakeholders and applied conservation (Driscol *et al.* 2012). When SDMs can bring together scientists, public stakeholders and policy makers, and are used as an adaptive management tool to understand complex landscapes that are undergoing changes, only then it has achieve its full potential (Polunin 2014).

Chapter 6: Effect of Fire and Grazing on Plant Community Composition and the Invasive Alien Species *Acacia nilotica* (L.) Willd. Ex Delile in Tropical Savanna at Bekol, Baluran National Park, Indonesia

ABSTRACT

Interactions between fire and grazing are important in determining when and how savanna shifts to another ecological state. However, although both factors are recognized as important drivers, not much research has been conducted to address the combined effect of these factors, especially on savanna community composition and woody invasive alien species (IAS), such as *Acacia nilotica*. Bekol Savanna in Baluran National Park, Indonesia, provides an ideal opportunity to: (1) determine the effects of fire, grazing and their interaction in shaping savanna plant community composition; (2) determine the effect of fire and grazing on *A. nilotica* seed germination and establish whether water buffalo play a role in the dispersal of their seeds; and (3) determine the response of *A. nilotica* to fire and its persistence in savanna ecosystems. Vegetation surveys along with examination of herbivore stools and soil seed banks and seed germination experiments were conducted to answer these objectives. Results showed the importance of interactive effect of fire and grazing on community composition in a tropical savanna ecosystem where it has also promoted the the invasive alien species *Acacia nilotic*a to establish in and dominate the community. Several characteristics of *A. nilotica* help the species become dominant, namely*:* ability to withstand fire and resprout following fire; ability to persist as a source of potential regeneration following disturbances in the form of an ample soil seed bank; and its high potential for endozoochory as demonstrated by its propensity to germinate in the seeds that remain inside buffalo stools.

Keywords: Fire and grazing, tropical savanna, *Acacia nilotica*, endozoochory.

6.1. INTRODUCTION

Fires in the tropics are widely believed to be important in creating and maintaining savanna (more open woody vegetation with grassy understory) in regions which, based on rainfall alone, should be forest (Bond *et al.* 2005). Sumardja and Kartawinata (1977) described savanna in Pangandaran Peninsula, West Java. They reported that Batu Meja and Karang Pandan Savanna sites in Pangandaran had been abandoned around 1957 and have not experienced any fire since then; subsequently, they have become young secondary forests dominated by *Decaspermum fruticosum*. Savannas located at high rainfall sites typically have at least some woody vegetation and in the absence of fire can have high tree cover that would outcompete and suppress grasses species, thereby disenabling further fires (Rosleine and Suzuki 2013; Scholes and Walker 1993). Thus, in these sites, regular fire is important to establish grass-tree coexistence (Sankaran *et al.* 2003). In contrast, savannas of drier sites, woody vegetation is mostly in low abundance and maybe absent, with grasses being the dominant component. Arid savannas have resources limitation (water competition) and lower biomass/fuels, and therefore fire is perhaps less influential for grass-tree coexistence in this type of savanna (Sankaran *et al.* 2005).

In savanna ecosystems, disturbance agents such as fire has been recognized as one of the key processes that can affect diversity levels and alter community composition and structure (Hobbs and Huenneke 1992). Fire events could catastrophically change the ecosystem over a short time scale (Scheffer *et al.* 2001). However the absent of fire may also hinder reproduction of certain plant species thus reducing species diversity in a given community over the longer term (Knuckey *et al.* 2016). Hence, these responses to fire initiates gradual readjustments in species interactions and ultimately composition through time in plant communities (Walker *et al.* 2007).

The size and life-stage of plants affects competitive hierarchies in savanna plant communities. In their juvenile stages, trees and woody shrubs are inhibited from emerging by interaction with grasses, but also fire, but once they have managed to overcome these obstacles and reach the canopy, trees can suppress grass understories which then results in reduced burning frequency (Bond and Wilgen 1996; Skowno *et al.* 1999). When fire has been absent for longer intervals, savannas are converted to non-flammable secondary forests or thickets which become increasingly protected from further fire intrusion with age (Hoffmann *et al.* 2012c). This is particularly so where adult woody plants can readily survive fire.

Fire survival generally depends on the following plant traits: thick bark, open crown architecture and ability to resprout from buds protected by bark or their subterranean position (Bond and Wilgen 1996; Marrinan *et al.* 2005). Certain woody plants, if the cambium is not damaged, can tolerate total defoliation by sprouting new shoots from epicormic buds, thus quickly regenerating the canopy. There are also plants with below-ground resprouting traits. Resprouting is a tolerance trait that confers persistence at the plant level, enabling it to survive diverse disturbance regimes. At the community level, this gives rise to biomes that are resilient to severe (biomass depleting) disturbance, including fire in savanna (Clarke *et al.* 2013). In some woody plants, buds proliferate at the bases of stems and form a woody swelling known as a lignotuber or burl (Bond and Wilgen 1996). In addition, there are also some species that characterize the fire-prone ecosystem that are not capable of resprouting and are killed by fire, and rely on mass recruitment after fire (obligate seeders). Meanwhile, other species are not killed by fire, but then die in the interval between fires. Nevertheless, their seed continues to exists in the community as dormant seed banks (above or below-ground) and they may re-appear after the next burn (Keeley *et al.* 2011). The cumulative number of seeds in the soil that are viable enough and have the potential to replace adult plants above ground is defined as soil seed bank (Alvarez-Aquino *et al.* 2005). Soil seed banks have a significant role in vegetation maintenance, succession, ecosystem restoration, invasive species management and conservation of genetic variability (Fowler 2012; Milberg 1995; Rogers and Hartemink 2000).

Herbivory, can also work to maintain savanna where woody plants are eaten and subsequently removed from the ecosystem (van Steenis 1972). However, it should be noted that where there are unpalatable woody plants and/or grasses are preferred by herbivores, grazing may promote woody cover in savanna, and this, together with lower fuel loads due to grazing, can result in less fires and a shift to a forest/woodland ecosystem. Subsequently, interactions between fire and grazing are important in determining when and how savanna shifts to another state (Sankaran *et al.* 2003; Scholes and Archer 1997). However, although both factors are recognized as important drivers, not much research has been conducted to address the combined effect of these factors (van Etten 2010). One of the few to study such combined effects is Staver *et al.* (2009) who studied the effect of the fire - herbivory interaction to suppress tree density in an African savanna. They found that combination of browsing and fire had much greater effects on tree density than either acting alone. Only fire and browsing together were able to prevent increases in tree density. This interplay led to the interesting phenomenon of retreat and advance of the savanna ecosystem, especially evident in by the movement of the sharp boundaries between savanna and forest areas (Desjardins *et al.* 1996; van Etten 2010). Fire and grazing are significant disturbances which lead to the expansion or retreat of grass-dominated ecosystems. Substantial research has focused on their effects as independent forces that alter the landscapes (Fuhlendorf *et al.* 2009). Previous studies only investigate these factors independently, often through small-scale homogenous experimental design units (Archibald 2008; Valone and Kelt 1999). Hence experiments or studies that decouple fire and grazing effects in savanna ecosystems are vital (Fuhlendorf *et al.* 2009).

Alien woody plant species, *Acacia nilotica* has been introduced to savanna in Baluran National Park in the late 1960s, where its original purpose was to create fire breaks to prevent fires spreading from Baluran Savanna to the adjacent teak forests (Sabarno 2002). However, since then, *A. nilotica* has spread rapidly and is now threatening the existence of Baluran Savanna as it has been observed to change the ecosystem structure from open savannas to thickets with a closed canopy of *A. nilotica* in some areas (Djufri 2004; Sutomo *et al.* 2016). In a northern Australian savanna ecosystems studied by Radford *et al.* (2001), *A. nilotica* was shown to have major negative impacts. Its adult trees are fire tolerant and can form thorny thicket formations (Burrows *et al.* 1986). This shift from open savanna to woodland and thicket has likely environmental impacts such as changes in herbaceous species composition, faunal habitat and soil hydrology (Radford *et al.* 2001a). Although adult trees of *A. nilotica* are fire tolerant, the effects of fire on juvenile plants of this species are lethal as Radford *et al.* (2001) showed high mortality of 1 and 2 years old seedlings of *A. nilotica*. Fire is seen as a useful tool for the arrest of *Acacia* spp. and woody thicket formation and also for the maintenance of open savannas in many parts of Africa (Radford *et al.* 2001b). The widespread occurrence of *A. nilotica* in the Baluran Savanna according to Tjitrosoedirdjo *et al*. (2013) may have been facilitated by large mammals such as the water buffalo (*Bubalus* sp.) that dominates the Bekol Savanna. According to them, at the end of the wet season and into the dry season, mature *A. nilotica* pods drop from the trees and are consumed by herbivores such as water buffalo. Tjitrosoedirdjo *et al*. (2013) suggested that the digestive system of the herbivores may scarify *A. nilotica* seeds, enhancing their germination, and that the herbivores would also facilitate the spread of *A. nilotica* in Baluran. It is expected that herbivores like buffalo do not browse on mature *A. nilotica* plants, although it is unknown for seedlings. Surprisingly, however, this proposal is lacking empirical evidence.

Therefore there is a need for an improved understanding of some key ecological traits, including likely responses to and recovery following fire and herbivore grazing, for this invasive alien species (IAS) to exist in savanna ecosystems, which will help determine the management options for the species in tropical savannas. Given the framework presented, the objectives of this chapter are to: (1) determine the effects of fire, grazing and their interaction in shaping savanna plant community composition in Bekol Savanna Baluran National Park; (2) to determine the effect of fire and grazing on *A. nilotica* seed germination and establish whether water buffalo play a role in the active dispersal of such seeds in the Bekol Savanna; and (3) to determine the response of *A. nilotica* to fire and its persistence in savanna ecosystem.

6.2. METHOD

6.2.1. Study Location

Baluran is located at the northern-most tip of East Java, on plains in the rain shadow of mountain ranges. Baluran National Park covers a vast area of 25,000 ha and is located in Situbondo District, East Java Province. On its northern side, it is bordered by the Madura Strait, and on eastern side is bordered by the Bali Strait. Baluran experiences a strong seasonal variation and a typical monsoon climate characterized by a long dry season. According to the Schmidt and Ferguson Classification, the Baluran

National Park belongs to climate type F, with rainfall of 900 - 1,600 mm/year, and air temperature of 27 - 34 °C (Caesariantika *et al.* 2011). Baluran National Park mainly consists of savanna (~40%), as well as dry forests (monsoon forest, secondary forest), primary forest, evergreen forest and mangrove forests (Figure 1).

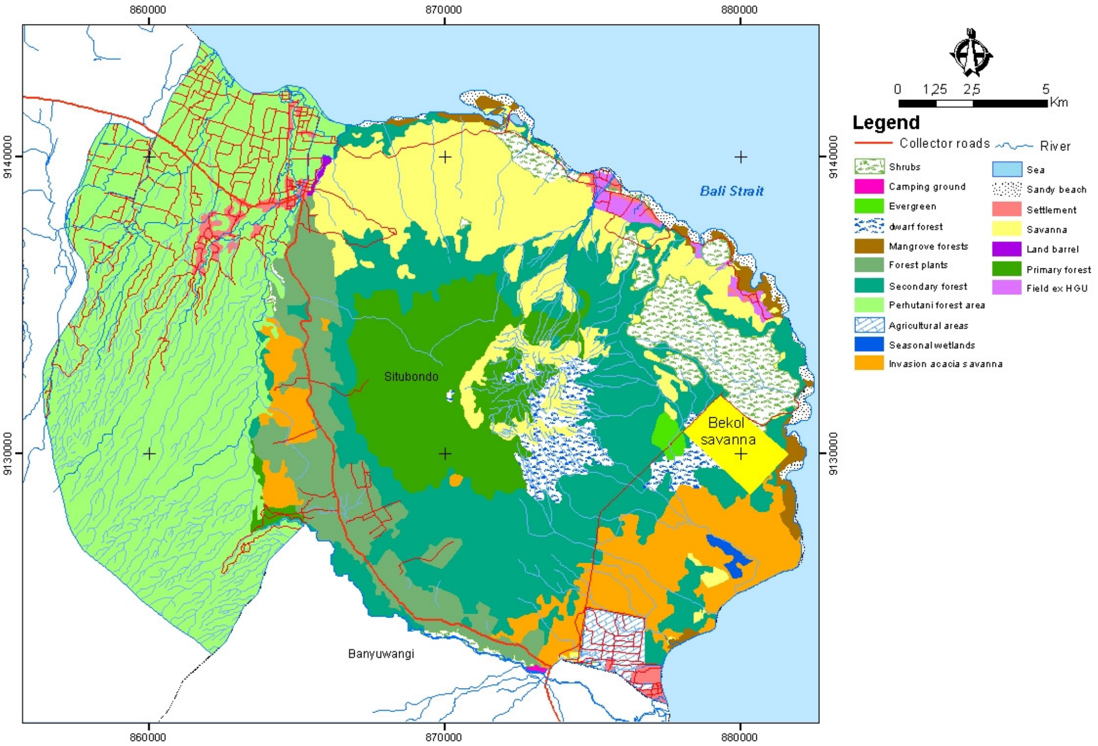


Figure 1. Map of location of the Baluran National Park in East Java Indonesia. Land use/landcover type is based on mapping provided by the Baluran National Park Management Office, 2013.

A. nilotica has spread rapidly in Baluran such as observed in Bekol Savanna (Barata 2000; Djufri 2004). The Bekol Savanna (bright yellow rectangle in figure 1), located in the eastern part of BNP, has a total area of 150 ha. Caesariantika *et al.* (2011) divided Bekol area into three types of vegetation structure: un-invaded area, invaded area, and managed area. The invaded area (~60 ha) is in the southern part of the savanna, and it has been invaded by *A. nilotica.* The managed area (~75 ha) is located in the central part of the savanna; it is actually also an invaded area but efforts to maintain this area as a savanna has been the priority of the BNP management by conducting controlled burning, as well as mechanical control almost every year, to remove *A. nilotica* from this landmark area of the BNP. An attempt to describe recent fire history of the Bekol area was done as part of in chapter 5 and shown in Figure 2.

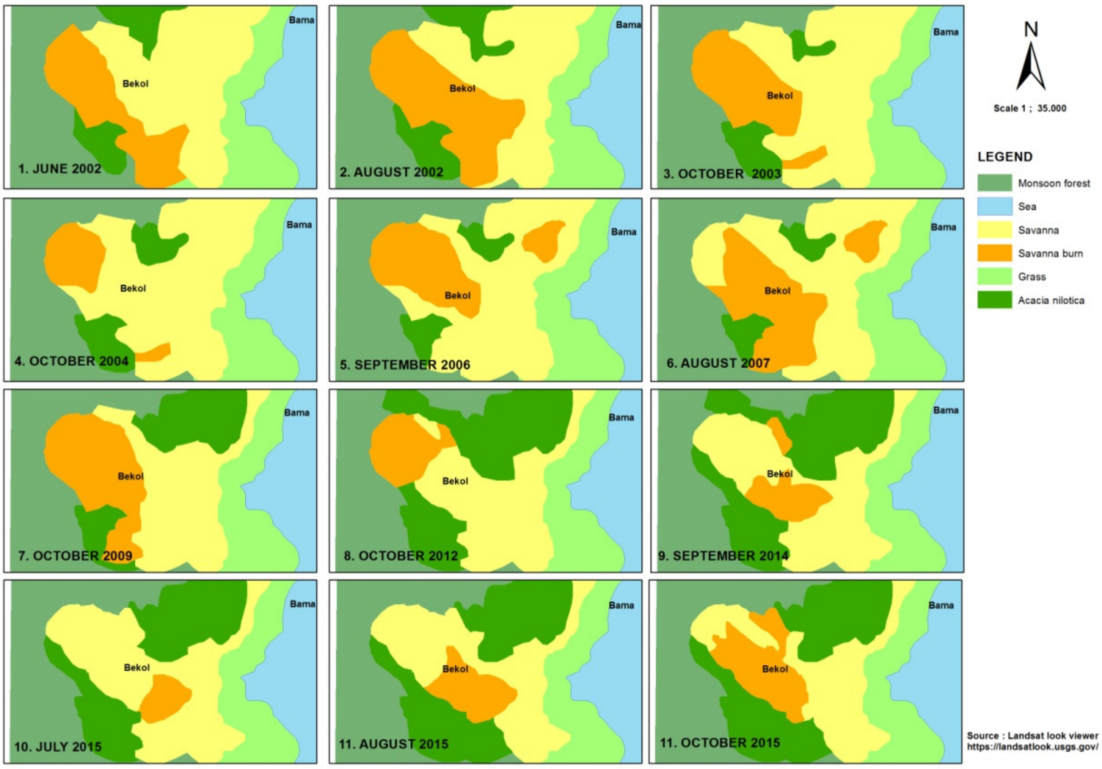


Figure 2. Map showing spatial and temporal patterns of *Acacia nilotica* invasion and recent fire in Bekol Savanna, Baluran National Park. Monsoon forest in legend refers also to Seasonally Dry Tropical Forest (SDTF).

6.2.2. Effects of Fire, Grazing and Fire Vs Grazing Interaction on Savanna Plant Community Composition

Fire history was acquired from various sources of unpublished reports, interviews with park managers and rangers, and also based on the map of fire history created in chapter 5 using remote sensing (Figure 2). In order to better understand the effect of fire, grazing and the fire - grazing interaction in shaping community composition specifically in Bekol Savanna, a sampling design using space for time substitution was employed. All sampling plots in Bekol area share similar climate, have similar volcanic origin soils, located in a plain (flat) topographical setting and experience similar fires event (burnt roughly at the same time of year under similar conditions).

I defined ‘fire’ (‘burnt)’ and ‘no fire’ (‘unburnt’) areas based on the typical fire regime of the BNP. Generally speaking, the fire area represent areas subjected to typical fire regime of the savanna (especially in terms of fire frequency, at least 3 fires per decade) whereas no fire were areas excluded from fire for at least ten years. I defined ‘grazed’ sites specifically for heavy grazing areas, whereas ‘ungrazed’ sites (light grazing) categories respectively based on the visual observations of grazing animals and also based on presence of their stools/dung as an indicator (Calviño-Cancela *et al.* 2006; Reitalu *et al.* 2010). Therefore ungrazed sites refer to those with no or very few scats and no observation of grazing during fieldwork. I also recognized that just because areas have a few or no scats and no grazing animals observed, it is likely that some grazing occurs given herbivores have free access to these areas. Ten areas of each combination were found and vegetation was sampled using quadrat plot (20 x 20 m). I measured plant abundance as density, a count of the numbers of individuals of a species within the quadrat plot (Endo *et al.* 2008; Kent and Coker 1992). Local plant names and scientific names, when known, were noted. Whenever there was any doubt about a species name, a herbarium sample was prepared. Drying and sample identification were carried out in Herbarium Baliensis in Bali Botanical Garden-Indonesian Institute of Sciences (LIPI). Vascular plant nomenclature was based on “Flora of Java” (Backer and van den Brink 1963) and names standardised to The Plant List (www.theplantlist.org). The effect of fire, grazing and fire-grazing interaction on savanna plant species composition was explored using permutation tests for multivariate analysis of similarity (PERMANOVA) available in PRIMER v.6. SIMPER (Similarity Percentage) analysis was then used to explore the relative contribution of individual species to average dissimilarity among groups, again using the PRIMER v.6 package (Clarke and Gorley 2005).

6.2.3. Effects of Fire and Grazing on *Acacia nilotica* Seed Germination and Dispersal

A). Effects of Fire and Grazing on *Acacia nilotica* Seed Germination

Random sampling points were selected from grid cells on a map of Bekol Savanna, Baluran National Park, and quadrats (2 x 2 m) were laid at each of these points, and a GPS was used to locate this point geographically. Buffalo stools in quadrats were then collected, put in a plastic bag and labelled; in total there were 30 stools found within quadrats. Similar steps were also applied for collecting mature seeds from pods growing on *Acacia nilotica* trees in *A. nilotica* stand areas (thickets). These stools and pods were then transported to Bali Botanical Garden. Twenty stools were broken up and *A. nilotica* seeds were removed by hand and also sieving. Seeds extracted from the crushed stools were then separated and only intact and undamaged seeds were selected for the germination test. As many as 519 seeds of *A. nilotica* were extracted from the 20 stools. However, only 310 of them were intact and undamaged.

A 2×2 complete factorial laboratory experiment arranged in a completely randomized block design, with 3 blocks as replications, was carried out in Seeds Laboratory at the Bali Botanic Garden from 27 November 2014 to 18 December 2015. The first factor was 'heat' (which was designed to simulate effects of a fire) with 2 levels: not heated (F0) and heated in oven with temperature of 105 °C for 5 minutes (F1), and the second factor was herbivory with 2 levels: seed extracted from dry pods (H0) and seed collected from buffalo stools (H1). Therefore, there were 4 treatment combinations, namely F0H0, F0H1, F1H0, and F1H1. Experimental units consisted of 10 seeds for each treatment combination and replicate, placed inside 10 cm Petri dishes lined with tap-watered tissue paper. Regular re-watering occurred as required during the course of the experiment.

In order to gauge seed imbibition, each seed in each experimental unit was weighed with an OhaussPioneer PA-2102 electronic scale every 1- 2 days starting from the intitial treatment application up until the end of the experiment. Seed which germinated was counted and excluded from the Petri dish. By the end of germination test, there were two kinds of count data gathered: count of imbibed seed – permeable seed which germinated, and therefore viable; and count of permeable seed which did not germinate, therefore non-viable. The seeds which did not imbibe, namely those which maintained a hard seed coat, were then scarified by cutting the seed coat in the area opposite the radicle with a sharp nail clipper. These seed were then tested for their viability by employing the tetrazolium test (Elias et al, 2012). The results of the TZ test were count data of impermeable seeds which were viable, and impermeable seeds which were non-viable. Generalised linear modelling (GLM) was conducted to analyse the data, using a Poisson distribution (for count data), a log link function and chi- square values based on likelihood ratios. GLMs were used to test the effects of the treatments (heat and herbivory, and their interaction) on the number of germinants, number of viable seed (V) and number of imbided seeds (I).

B). *Acacia nilotica* Seed Dispersal

The remaining ten stools from the sample described above (A) were also broken up and mixed together and then the numbers of seeds found in them were noted. There were 115 seeds inside these stools. Seeds were kept in the broken up stools for observation of any emerging seeds coming out from the stools (germination directly from stools treatment), with regular watering during the course of the experiment. Germination beds were prepared under a shade of a paranet (shadecloth) in the nursery house of Bali Botanical Garden. The planting media composed of soil, humus and sand in equal proportions (1:1:1) by volume. Then 30 seeds extracted from dry pods (collected as per the 6.2.3.A. sample) were planted in three rows on the beds (three replicates), this acted as 'control' or no treatment. Regular watering occurred as required during the course of the experiment. Germination percentage was calculated and subjected to analysis of variance (ANOVA) test.

In addition to seed germination tests, I compared *A. nilotica* seed dispersal by various herbivores in the field by recording the position (with a GPS) every time I encountered *Cervus timorensis, Bubalus* sp. individuals and their stools in several locations of Baluran National Park, including the Bekol area. I noted the number of stools from each animal and the amount of *A. nilotica* seeds inside every stool. I also noted the distance from the stools with the nearest mature *Acacia nilotica* parent individual.

6.2.4. Response of *Acacia nilotica* to fire and its persistence in savanna

A). Resprouting

Sampling was conducted in the rainy season, four months after the September 2014 fire. During the rainy season in November and December 2014 Baluran had approximately 59.6 mm of rainfall (BPS 2014). The rainy season in Baluran generally runs from November to April. Sampling was conducted in January 2015 to ensure that all or most germinable seed would have germinated. Random point sampling was applied in the Bekol Savanna area that was burned in September 2014. Eight random points selected on map and then these were located on ground using a GPS. In each of these locations, sampling plots 50 x 50 m in size was established. All *A. nilotica* plants in plot were counted and measured for their girth (stem diameter or dbh) and height and assessed for evidence of resprouting and the type of resprouting. I used the classification of resprouting type by Marrinan *et al.* (2005) and divided the resprouting into two types: (1) basal resprout (original stem damaged/one or more shoots sprouting from the base of the damaged stem); and (2) above-ground resprout/stem resprouting (original stem damaged/one or more shoots sprouting from along the original damaged stem). As many as 350 individuals of *A. nilotica* were sampled across the eight plots. Besides resprouting observations, I also noted the number of *A. nilotica* seedlings that emerged following the fire in each plot.

B). Soil Seed Bank

Sampling for the soil seed bank (SSB) was conducted in four different sites (all in Bekol area) namely: recently burned site, grazed site, restoration site and border site. The burned site was site that was burned in August 2014 (4 months before sampling). The grazed site was an area with many grazing animals as indicated by abundant presence of their stools/dung, but not burnt for at least ten years. The restoration site was where, since 2010, Baluran National Park management has been applying eradication program through poisoning and slashing of the *A. nilotica*. Lastly, the border site was located at the boundary between the savanna and the *A. nilotica* stand, from 50 m into the interior (*A. nilotica* stand) to 50 m into the exterior (savanna). Soil samples were collected along five 50 m transects at each site (with border transects placed perpendicular with vegetation boundary). Along each transect, five small quadrats of 20 cm x 20 cm were randomly made and soil sampling was conducted using iron frame (same size as quadrats) to 15 cm depth.

In the laboratory, each soil sample was air dried to prevent germination before hand-sorting of the samples. Extraction of seeds in the soil followed Aparecida & Miranda (2014) and involved breaking up soil clods and then sieving. The total number of *A. nilotica* seeds in soil samples was estimated as the sum of seeds visually detected (direct counting). This is likely to be accurate as *A. nilotica* seeds are relatively large (7 mm to 1 cm). Survival of seeds in the soil does not depend merely on avoiding germination, but viability also must be preserved (Mansur 2011). Most seeds in arid habitats and savanna show orthodox seeds storage behaviour (Murdoch and Ellis 2000). On the other hand, embryo viability is the other important aspect of seed germination. Many procedures to perform viability test, including conventional methods and quick test such as tetrazolium test, have been employed widely in a variety of contexts (Elias and Garay 2004). *Acacia nilotica* seeds that were intact (not obviously damaged) were randomly selected for tetrazolium test. For burnt plots, five seeds were tested, in grazed site 51 seeds, restoration site 12 seeds and in border site 41 seeds (numbers varied in line with numbers collected from each site). Tetrazolium (2,3,5-triphenol-tetrazolium chloride) is a colourless solution having a reddish-pink colour that stains living tissue, such as live embryos. To do this test, seeds were first soaked in water for one night, cut to facilitate stain intake and incubated in a 2% solution of 2,3,5-triphenol-tetrazolium chloride in phosphate buffer for 24 hours in the dark (Calviño-Cancela et al. 2006; Elias and Garay 2004; Peters 2000). Those stained red had living embryos and were scored as ‘viable’.

6.3. RESULTS

6.3.1. Effects of Fire and Grazing on Savanna Plant Community Composition

PERMANOVA resulted in a close to significant ($P_{permutation}$ = 0.068, > 0.05) interaction between fire and grazing (Table 1). There is also a significant effect of both fire and grazing on their own. *Acacia nilotica*, an invasive alien species, was present in all four groups (no fire, fire, no grazing, grazing), but has higher average abundance at both fire and grazing sites compared to other species (Table 2). *Azadirachta indica* and *Desmodium* sp. (local species) were also present in all four groups but, they were less abundant in the fire group and grazing group, indicating the reverse condition to *A. nilotica*.

Table 1. Result from PERMANOVA analysis.

Source	**df**	**SS**	**MS**	**Pseudo-F**	**P (perm)**	**Unique perms**
Fire	1	9224.4	9224.4	3.7336	0.001	998
Grazing	1	8176.4	8176.4	3.3094	0.007	998
Fire x Grazing	1	4788.7	4788.7	1.9382	0.068	998
Res	30	74120	2470.7			
Total	33	99660				

Table 2. SIMPER analysis results showing species contributing most to the dissimilarity between site groups and their average abundances (sq root transformed cover).

Species	**Life form**	**No Fire**	**Fire**	**Contrib %**	**No Grazing**	**Grazing**	**Contrib %**
		Av. Abund.	**Av. Abund.**		**Av. Abund.**	**Av. Abund.**	
Acacia nilotica	Tree	1.31	2.05	16.48	0.78	2.43	23.75
Achyranthes bidentata	Herb.	0.87	0	5.29	0	0	0
Azadirachta indica	Tree	2.42	0.17	12.26	1.64	0.71	7.36
Desmodium sp.	Herb.	1.73	1.1	9.61	2.73	0.4	17.74

Dichanthium coricosum	Grass	2.83	0.11	18.23	0.92	1.45	5.6
Ocimum sp.	Herb.	0.44	0.22	4.77	0	0.52	3.24
Passiflora foetida	Herb.	0	0	0	0.76	0.07	7.98
Polytrias amaura	Grass	2.06	0.94	11.94	1.64	1.24	10.18
Thespesia lampas	Herb	1.17	0.87	7.57	1.09	0.93	8.98
Ziziphus mauritiana	Tree	0	0	0	0	0.49	3.29

6.3.2. Effects of Heat & Grazing on *Acacia nilotica* Seed Germination and Dispersal

A). Effects of Heat and Grazing on *Acacia nilotica* Seeds Germination

Table 3 shows result from the GLM analysis. In general the results showed that grazing (herbivory) does not effect viability of seeds, but 'fire' (heat treatment) does. Both herbivory and heat have a slight (but significant) negative effect on germination (i.e. more seed germinated when no treatments applied). There was a significant interaction between herbivory and heat on imbibition, with more heated seeds imbibing (i.e. permeable) after consumption by herbivores than when not eaten. Germination rates are relatively low (overall 8.3%) compared to viability (overall 46.7%).

Table 3. Generalised Linear Modelling of germination experiment – testing of effects based on number of germinants. Probability Distribution: Poisson. Link Function: Log. Scale Parameter Method: Deviance. Chi square values based on likelihood ratios.

Variable	**Herbivory**			**Heat**			**Heat X Herbivory**		
	Chi-Sq	**p-value**	**Effect**	**Chi-Sq**	**p-value**	**Effect**	**Chi-Sq**	**p-value**	**Effect**
Germination	9.07	0.03	-ve	9.07	0.03	-ve	0.80	0.40	none
Imbibition	0.32	0.57	none	0.32	0.57	none	4.38	0.036	+ve*
Viability	0.68	0.41	none	8.98	0.03	-ve	2.56	0.11	none

The existence of buffalo stools on a seed appears to offer protection from fire. It can be seen on the interaction graphic (Figure 3) that percentage of impermeable seed which are non viable for seeds from buffalo stools are significantly less than of seeds from direct pods extractions counterpart.

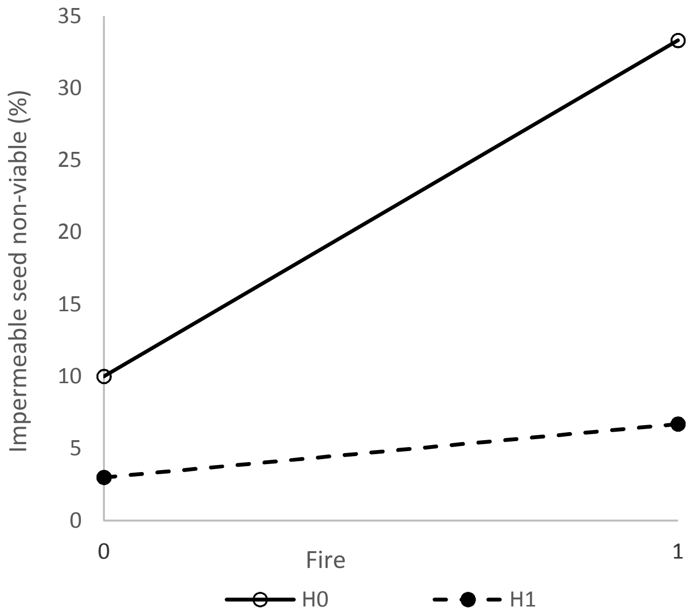


Figure 3. Interaction graphic between heat (F) and herbivory (H) on IN

B). *Acacia nilotica* Seeds Dispersal

Germination of field collected seed started to occur on the 25th day of observation. Measurements of seedling length on the treated (inside buffalo stools) and control seeds (not consumed), revealed that seedling grew more quickly after germinating in the buffalo stools compared to control seeds sown in a planting media (Figure 4). Between days 25 to 35 was the period of fastest seedling growth for both control and stool seedlings. In general, germination percentage was highest in the seeds that remain inside stools compared to that of control treatments ($p<0.05$) (Table 4).

Table 4. Differences in germination percentage of *Acacia nilotica* in the treatment and control. Superscript letters (a, b) after mean values (±SD) indicate significant differences between sites as assessed with Tukey's^{B} test.

Treatment	Germination percentage
Control	2.22 (± 1.11)^{a}
Stools	10.98 (± 2.92)^{b}

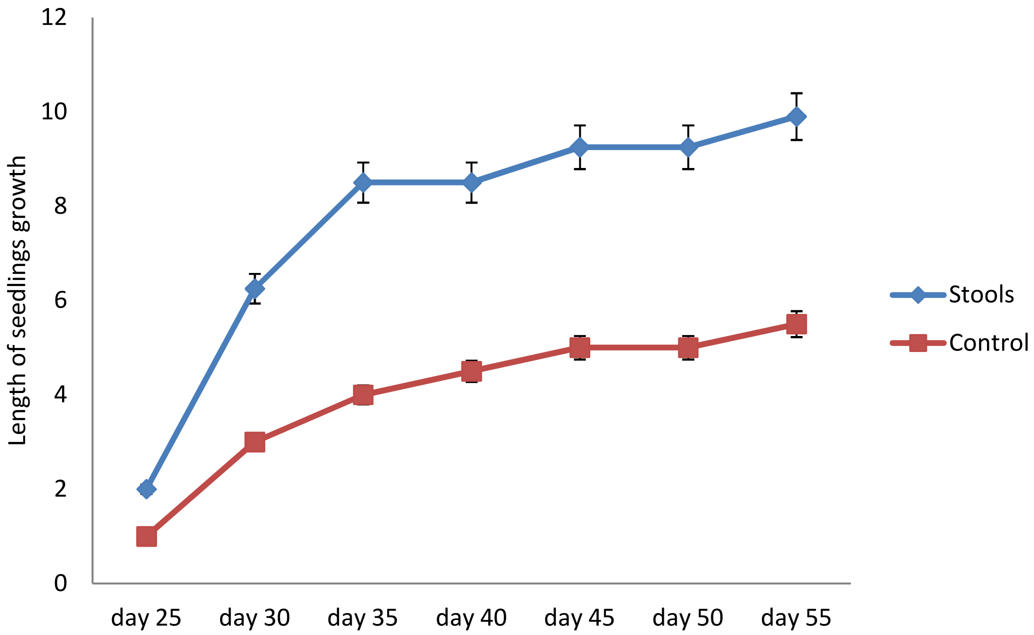


Figure 4. Mean length of *Acacia nilotica* seedlings growth (cm) in the stools treatment and control. Stools treatment refers to *A. nilotica* seeds that germinate directly from inside buffalo stools whereas control refers to *A. nilotica* extracted from fallen pods from *A. nilotica* mature trees.

Figure 5 shows comparison in terms of *Acacia nilotica* seeds dispersal by other herbivores in Baluran National Park. Barking deer (*Cervus timorensis*) stools were much higher in amount compared to stools of buffalo (*Bubalus* sp.). However, buffalo stools contain highest mean number of seeds of *A. nilotica* inside them compared to barking deer. Meanwhile in terms of mean distance between the stools with mature *A. nilotica* and herbivores, the barking deer stools was commonly found further apart from the mature *A. nilotica* whereas buffalo stools were generally found nearest to mature individual of *A. nilotica.*

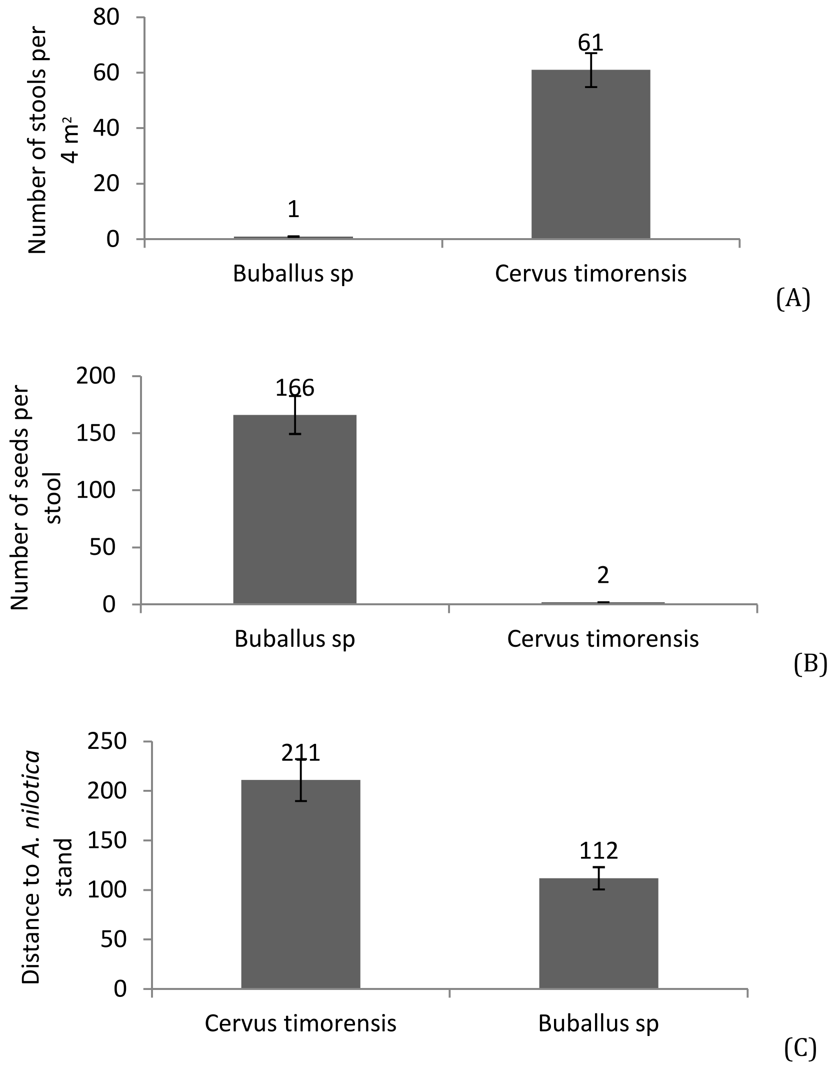
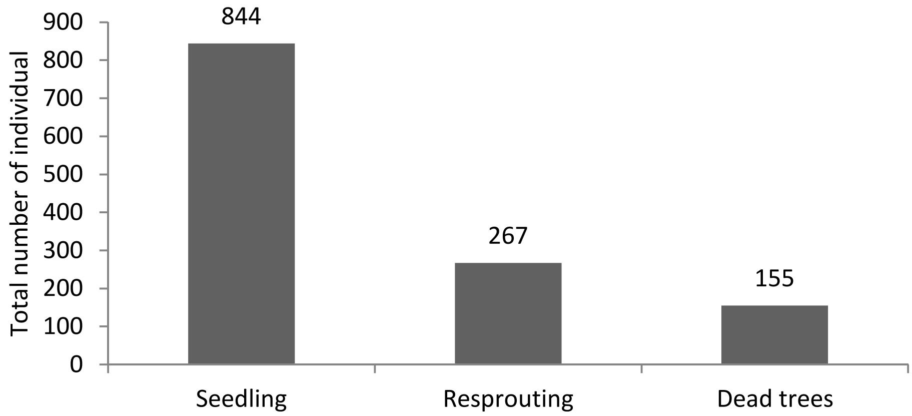


Figure 5. (A) Mean number of stools among different herbivores in Baluran NP. (B) Mean number of *Acacia nilotica* seeds found in the stools from different herbivores. (C) Mean distance (metres) of stools among different herbivores from nearest mature *Acacia nilotica*.

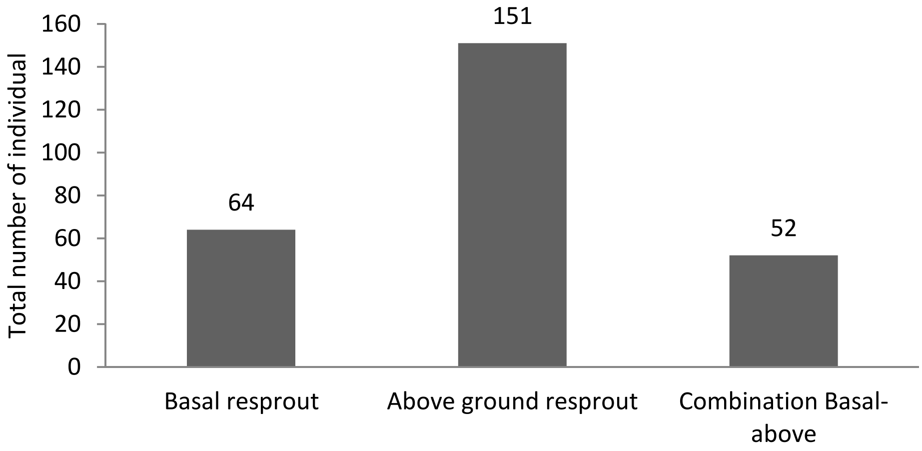
6.3.3. Response of *Acacia nilotica* to fire and its persistence in savanna

A). Resprouting

It was confirmed in the field that *Acacia nilotica* is able to regenerate following fire by seedlings and resprouting as both showed strong response (Figure 6). As many as 844 seedlings of *A. nilotica* were observed (0.0422 seedlings per m^2), whereas 267 resprouting plant were found (0.0133 trees per m^2). Above ground/stem resprouting was more prominent (56.5%) compare to basal (23.9%) and combination of basal and stem resprout (19.4%).



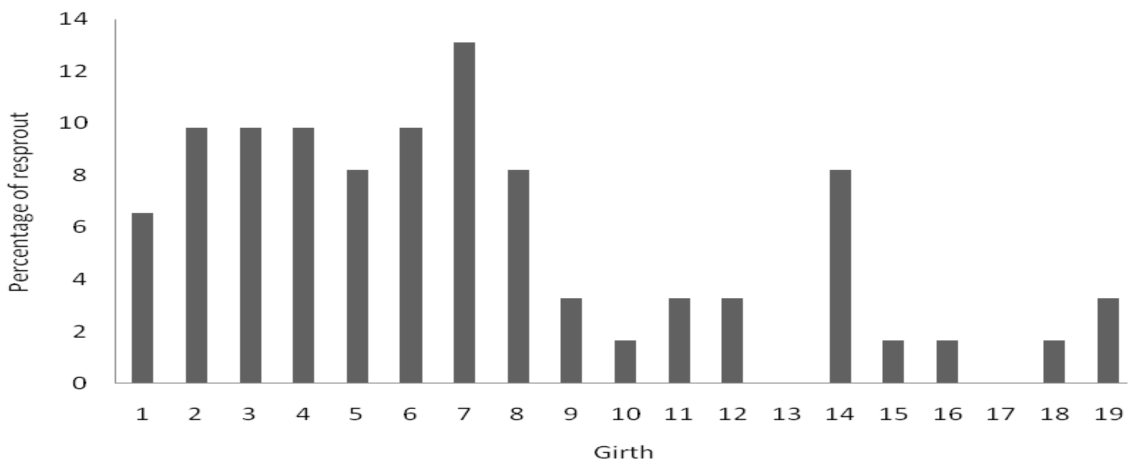
(A)



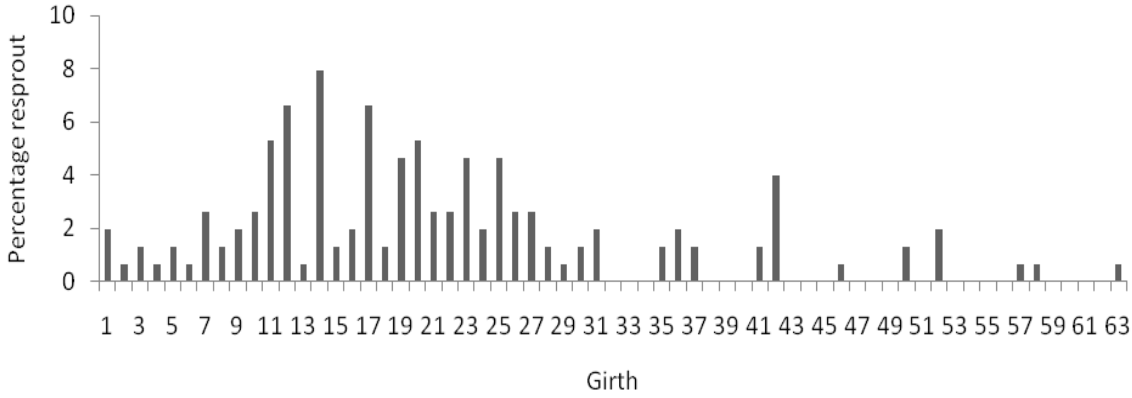
(B)

Figure 6. Total numbers of *A. nilotica* seedlings and resprouting plants across all plots (A). Comparison of total number of *A. nilotica* individuals which showed basal, above-ground and combination resprouting across all plots (B). Plots = 38

Highest percentage of basal resprout was at 7 cm (girth) *A. nilotica* (more of this sized plant in my sampled population) (Figure 7a). Above-ground resprouting was found in *A. nilotica* across wider range of girth than basal resprouting (Figure 7b). However, both basal and above-ground resprouting can occur in saplings with stem girth as low as 1 cm in diameter. However, the maximum percentage of above ground/stem resprout was shown at 14 cm of girth size.



(A)



(B)

Figure 7. Percentages of total basal resprouting individuals of *A. nilotica* for each 1cm stem girth class (A). Percentages of total above-ground resprouting individuals of *A. nilotica* for each 1cm stem girth class (B). Note" girth here refers to dbh (cm).

B). Soil Seed Bank

In terms of soil seed bank (SSB), the burned site has the lowest average seed density, whereas the border site retains the highest mean number of seeds (Table 5). There was no difference in terms of mean number of seeds in SSB between the burned site and restoration site, and between the grazed and border sites. Figure 5 shows the viability percentage (based on TZ test) of the SSB at different sites. The burned site retains the highest SSB viability percentage (80%), which was close to that found at the grazed site (78%), followed by restoration site (60%), with the border site having the lowest viability rate (45.6%). The interior (*A. nilotica* stand) and edge have far higher percentage of seeds in the SSB when compared with the exterior part (savanna) (Figure 9). It also shows that those savannas adjacent to *A. nilotica* advancing edge had the highest SSB.

Table 5. Differences in mean number of seeds in the soil seed bank (SSB) between sites. Superscript letters (a-b-c) after mean values (±SD) indicate significant differences between sites as assessed with Tukey's[B] test.

SSB Location	Mean number of Seeds (per 400 cm^2) in SSB
Burned site	1.73 (± 0.56)^a
Grazed site	11.00 (± 3.67)^b
Restoration site	5.5 (± 2.01)^a
Border site	11.2 (± 4.7)^b

Figure 8. Viability percentage (TZ test) of soil seed bank found in different sampling location in Bekol Savanna Baluran National Park. Burnt site N = 5, grazed N = 51, restoration N = 12, border N = 41.

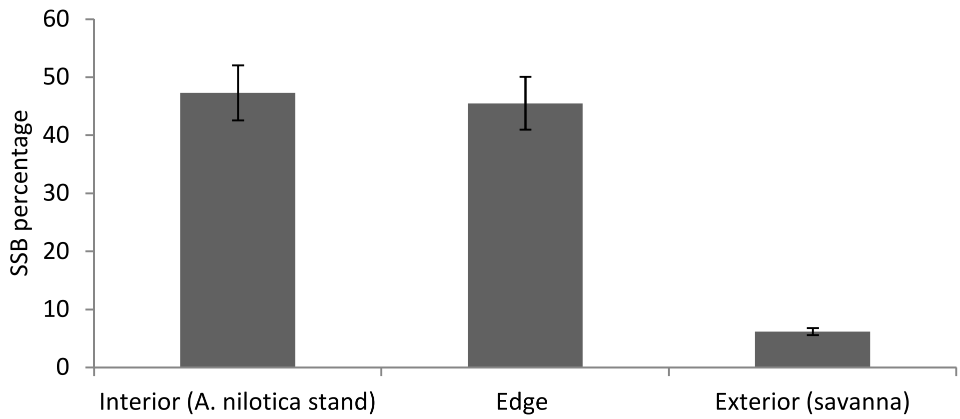


Figure 9. Percentage of soil seed bank (SSB) found in interior (*Acacia nilotica* stand), edge and exterior (savanna) in Bekol, Baluran National Park. Values are % the total of all seeds found in soil samples (as a measure of relative abundance).

6.4. DISCUSSION

At the landscape scale, fire in the Bekol Savanna within Baluran National Park (BNP) influenced plant community composition, as did grazing and, most likely, interactions between grazing and fire. In this savanna, where prescribed burning to create forage in grasslands and savannas for certain wildlife occurs, there is some evidence that the combined effects of fire and grazing influenced floristics (high species turnover between sample groups with very few species in common). Several plant species appear to be impacted by fire or grazing (mostly negatively), whereas some appear affected by the interaction of these factors; however, more study is required to make more definite conclusions on the responses of specific species.

Grazing is prominent in BNP. The BNP is home to wild deer, wild Java Buffalo, and wild water buffalo as its grazing animals, but grazing intensity varies across the landscape. Baluran National Park managers regularly apply controlled burning to reduce its fuel load and also provide fresh grasses and other palatable species for the grazing animals, although wildfires also occur, creating a mosaic of different burnt ages across the landscapes. Fire and grazing are prominent disturbances in the formation and maintenance of savanna and grassland (van Langevelde *et al.* 2003). As mention earlier, despite the importance of fire and grazing as determinant factors, only a few studies addressed their combined effect (van Etten 2010). Previous study have mostly investigated these factors independently, sometimes through small, homogenous experimental units (Archibald 2008; Valone and Kelt 1999). This study supports research attempting to decouple fire and grazing effects in savanna and grassland landscapes (Fuhlendorf *et al.* 2009). Fuhlendorf *et al.* (2009) introduced the concept of pyric herbivory which literally means grazing driven by fire. They show that the interaction between free-roaming bison (*Bison bison*) and random fires encourage heterogeneity and provides the basis for biological diversity and ecosystem function of North American and African grasslands. The individual and combined effects of fire and grazing influence the ecological structure and functioning of many ecosystems as fire and grazing are dynamic processes that interact with each other and with spatial and temporal variability across complex landscapes (Fuhlendorf *et al.* 2009).

Although controlled experiments which are regularly monitored are perhaps the most common way to study interactions, using a space-for-time substitution approach does have some advantages, such as to working at larger scales rather than small experimental plots. Using space for time (as used in this study) has certain requirements, such as its assumption that only the difference between sites is their fire and grazing history. This requirement has been met in this study, as explained earlier in the methods section of this chapter, as well as in the results section in Chapter 4. Indeed, in order for the space-for-time substitution approach to be useful in inferring patterns of vegetation change, firstly, the environmental conditions such as the climate, substrates and topographical positions of the sites under study need to be less varied between groups than within groups (Durán *et al.* 2009; Johnson and Miyanishi 2008; Pickett 1989). Secondly, sites within groups need same disturbance history and have similar pre-disturbance conditions. When these assumptions are met then the space-for-time substitution approach can be a powerful tool (Durán *et al.* 2009; Johnson and Miyanishi 2008; Wills 2002; Zhu *et al.* 2009). A space-for-time substitution approach has been applied and used in a wide range of studies to determine the effects of fire (Gonzalez-Tagle *et al.* 2008; Watson and Wardel-Johnson 2004) and land abandonment (Baniya *et al.* 2009; Myster and Pickett 1992a; b). The approach has also been used to examine the dynamics of vegetation (Wills 2002; Zhu *et al.* 2009), soil nutrient status and biomass (Bautista-Cruz and del Castillo 2005; Bormann *et al.* 1995; Durán *et al.* 2009; Wang *et al.* 2009) and forest structure (Spencer *et al.* 2001).

One of the main results of this study in terms of plant community composition in Bekol Savanna is the domination of *Acacia nilotica*. *Acacia nilotica*, an invasive alien species, was present in all four groups (no fire, fire, no grazing, grazing) but it shows higher average abundance in both the fire and grazing groups when compared to other species. This indicates that this species is both somewhat fire tolerant and unpalatable. Indeed, most other common woody species are negatively affected by fire and/or grazing, *A. nilotica* is one of few to be promoted by fire and grazing. It has been confirmed in this study that *A. nilotica* pods are grazed by buffalo which spread the seeds to many areas of savanna (through the seeds that remain in their stools/dung) including areas which subsequently experience fire. Once it has experienced fire, the heat may have assisted in scarification process of the seeds and sped up the germination, especially where seeds are buried in soil or protected inside stools. When there was no subsequent medium to high intensity fires for several years, these seedlings may grow to maturity becoming dominant at these sites and becoming resistant to future fires by resprouting.

For biological invasion to occur, a definite seed dispersal capability is required. Many weeds possess well-adapted accessories to aid in long-distance movement of their seed (Radosevich *et al.* 2007). Endozoochory (seeds carried internally by animals via ingestion) is most common among vertebrates. To be considered as effective seed dispersal agents, the animals have to supply some benefits to the dispersed seeds, such as long distance dispersal into new and accommodating environments (Comins *et al.* 1980). In the Serengeti, and in tropical savanna generally, ecological processes, including seed dispersal, are dominated by large herbivores ((Anderson *et al.* 2014).

Acacia nilotica is unpalatable to most herbivores as this plant possess thorny spikes on branches makes it difficult for the herbivores to consume the leaf. The pods that drop to the ground, however, are usually consumed by herbivores, especially during prolonged dry periods when there is limited availability of fresh shoots and grasses. At the end of the wet season and into the dry season, mature *A. nilotica* pods drop from the trees and are consumed by herbivores such as water buffalo (Tjitrosoedirdjo *et al.* 2013). This leads then to the dispersal of this species throughout the national park. My results show high numbers of undamaged *A. nilotica* seeds inside buffalo stools, a high percentage viability of these seeds, and a clear tendency for seeds to germinate directly from inside the stools. Sabarno (2002) found 26 – 45 seeds of *A. nilotica* in water buffalo stools, ~62 in wild Java Cattle, known as Javan b*anteng* (*Bos javanicus* subsp*. javanicus*), and ~9 seeds in deer stools. This pattern is in line with my findings however in my results I found fewer seeds in deer stools. This shows the high probability of *A. nilotica* dispersal by these herbivores, especially water buffalo, as banteng are now rarely seen in the wild and is categorized as an endangered species on the IUCN Red List (IUCN 2014; Suhadi 2009; Widayanti 2010). I presume that the amount of seeds found inside stools will vary between seasons. The number of seeds found inside stools may be higher in the dry season when herbivores opt to consume *A. nilotica* pods as result of limited availability of fresh shoots and grasses and, vice versa, the numbers will be lower in the wet season. However it is likely that buffalo is the most common mode of *A. nilotica* seed dispersal in BNP.

Buffalo stools might act as an important microsite that protect *A. nilotica* seeds and promote their germination. Firstly they may provide a distinct microclimate of humid conditions and a shelter from the extreme air temperatures at Bekol Savanna in BNP that can reach 40°C during dry season. Moisture retention by stools may are likely to play an important role in seed germination. The importance of microsite had been recognized as an important element in plant invasion and succession, especially in primary succession in volcanics environment (Gomez-Romero *et al.* 2006; Mori *et al.* 2008; Titus and del Moral 1998). In this case, microsite provided microclimatic and substrate amelioration. In a study of plant colonization following eruption of Mt. St. Helen, Titus and del Moral (1998) found that colonization patterns of plants in microclimates were changing over the years which highlighted the dynamic nature of the landscape and significance of climate, substrate amelioration and seed rain to plant establishment. Availability of microsites for overcoming seed dormancy are essential for increasing the likelihood of seed germination and seedling establishment, particularly in dry environments (Abari *et al.* 2012). These results also further confirmed the important roles of seed dispersal via passage through animal guts or endozoochory. By studying droppings/stools of emu (*Dromaius novae hollandiae*) in three different locations in Western Australia, Calviño-Cancela *et al*. (2006) found emu to be an important non-standard dispersal agent for long-distance plant dispersal. In other studies, Calvino-Cancela and Rubido-Bará (2012) found that seeds having been through the guts of slugs escaped from physical damage and the ingestion helped seed scarification and sped up its germination. In another ecosystem, a volcanic terrain in Mount Koma in Japan, Nomura and Tsuyuzaki (2015) reported that hares were the agent of dispersal and also aided the establishment of seedlings following volcanic eruption.

The importance influence of grazing (herbivory) and fire in the process of expansion or retreat of one ecosystem over another (savanna or *A. nilotica* stand) in Bekol is supported by the results of my seeds experiment. The results confirmed that heat treatment (designed to simulate the effect of fire) slightly reduced seed viability and germination; this was probably because heat treatment of 105° C for 10 minutes is more than would be experienced in a fire especially if seeds are in the soil. The experiment also showed that the combination of heat and herbivory had the highest amount seed coat cracking, probably because of the insulating effect of stool remains around seeds. In general, heat had a detrimental effect on seed viability. However the existence of stools at the surface of seed coat seems to protect seeds from detrimental effect of heat. The results also suggested that buffalo grazing and ingestion might not alone have sufficient scarification effect on the seeds that would speed up the germination; fire may have played this role but appears limited to those seeds with additional protection layer such as the buffalo stools. Stools from buffalo grazing provide a protection layer which prevents overheating from fire which may damage the seed. Further ample germination of seed was observed in the study area in the wet season after a fire, indicating that soil also provides insulation from heat. Radford *et al*. (2001) found that germination was higher in scarified seeds, indicating that the hard seed coats of *A. nilotica* seeds inhibit germination. Overall, germination rates are relatively low compared to viability suggesting more is study required (more treatments, more replicates) to determine best combination of treatments to germinate seed. Hardened testa (seed coat) is a common phenomenon reported for Fabaceae, including *A. nilotica*, which inhibits germination (Radford *et al.* 2001a). Impermeability of the *A. nilotica* seed coat prevents water uptake (imbibition) which is the first step in its germination. In other words, such seed has physical dormancy (Baskin and Baskin 2014). Moreover, Baskin & Baskin (2014) explained that water-gaps or water plugs in the seed must be removed in order to germinate. Artificially, it can be done by mechanical or chemical scarification (Hilhorst 2007) and naturally by mechanical impacts, fire (Radford *et al.* 2001a), seed coat decay and possibly other disturbances.

I have shown that adult plants and saplings of *A. nilotica* are able to withstand fire. Following fires in Bekol Savanna, *A. nilotica* demonstrates a prominent vegetative response*. Acacia nilotica* shows resprouting ability following fires, with above ground/stem resprout being the most common type. The ability of species to resprout following defoliation differs with age, size and fire severity (Hobbs and Mooney 1985). Some species only have the capacity to sprout when they are young as shown in many leguminous savanna trees (Bond and Wilgen 1996). However, Radford *et al.* (2001) shows 1 and 2 years old seedlings of *A. nilotica* are very vulnerable to even mild fire; however I found that even small seedlings of 1-2 cm stem diameter can resprout. There has been reported impact of frequent fires on resprouting species (Knuckey *et al.* 2016). Generally, plants produce new epicormic buds at a slow rate, so that very frequent fire can lead to all of them being used (Knuckey *et al.* 2016). Resprouting is now recognized as a key functional trait among woody plants and as the basis for the persistence niche and Clarke *et al*., (2013) have developed the *BPR* or buds-protection-resources scheme to understand resprouting in fire-prone ecosystem (Clarke *et al.* 2013).

In terms of the soil seed bank (SSB), the grazed site and border site (between savanna and *A. nilotica* stand) in Bekol area, Baluran National Park, have the largest SSB of *A. nilotica*. At the border site, a large SSB in the interior (*A. nilotica* stand) may be caused by the high abundance of pods that fell off the *A.nilotica* trees in the area. Savanna sites being actively grazed by herbivores and those adjacent to the *A. nilotica* advancing edge had the next largest SSB. Over the long dry season, the felled pods were consumed by buffalo or other animals near the border site, and when these herbivores move to water ponds and grasses in the savanna to cool off and graze, the consumed seeds were dispersed in their stools where they defecated. Old stools would break down over time and eventually blend with the soils. These seeds are then potentially stored in the soil as a persistent seed bank. This may have caused the high number of seeds in the SSB of the grazed site. Beside cattle, elsewhere in the world, particularly in Australia, ants play a major role in local dispersal of Acacia seed (Davidson and Morton 1984; Holmes 1990; Montesinos *et al.* 2012). Although this was not observed in Baluran, there are numerous ants which potentially play this role, and it is an area recommended for future research. The possible reason for the small number of SSB found in the burned site is perhaps due to high mortality of the seeds that was burned in the fire events or had germinated and died or were now seedlings, depending on when the fire occurred and if there has been rain since then. Acacia seeds generally survive fire when in soil, therefore perhaps only the surface ones were killed. Some seeds of the SSB may also have already germinated following the fires. Radford *et al.* (2001b) found that the mortality rate following fire for *A. nilotica* seeds that were buried 1 cm from the surface were lower than the seeds inside cattle stools. They conclude that fire may act as a scarifying agent for seeds that are not killed outright. Although the amount of SSB was low in the burned site, the viability percentage here was the highest.

The restoration site has moderate amounts of *A. nilotica* seed in the soil. Starting in 2011, eradication efforts are led by the Centre for Research and Development of Forestry Ministry (PUSLITBANGHUT). Basically the method is to apply a strong herbicide (triclopyr solution mix with diesel) to the bark of *A. nilotica* (stem brushing) (Sutomo 2014). After one year, the leaves begin to fall off and the bark and stem tissuedecays, eventually causing the tree to break and fall. Soon after the trees will be burned and planting of grass will begin. Although the restoration was successful at the site studied, there was some persistent SSB. Therefore, although the above control methods seem quite promising, it still needs to be assessed for its effectiveness, especially when taking in account the potential for regeneration via the soil seed bank.

Several biological characters are known to facilitate invasion by *A. nilotica* into the Bekol savanna. Adult plants of *A. nilotica* are apparently able to withstand fire and also prolonged drought period (Radford *et al.* 2001a; Sabarno 2002) which are both common in Baluran National Park (Sabarno 2002). In the field, even quite small (and presumably young) *A. nilotica* affected by fire in Bekol Savanna were also observed to readily resprout following recent fire. The ability to withstand fire perhaps was one of the causes of the domination of *A. nilotica* in the sampling sites of our study. Very high density of *A. nilotica* was also reported by Setiabudi *et al.* (2013) where it was estimated that the density reached approximately 2000 trees per hectare comprised mainly of small trees of *A. nilotica.* Elsewhere, there are also other reports that show increased density of *A. nilotica* after fire (Pratt and Knight 1971; Radford *et al.* 2001a). *Acacia nilotica* can form a dense thicket formation, which when combined with its fire resistance (mass seedling and resprouting) would further enhance the invasiveness of the species.

In Australia, impact of *A. nilotica* on Mitchell grasslands was studied by Burrows *et al*. (1986) where it was observed that *A. nilotica* was invading the grasslands and forming thorny thicket formations. In Baluran National Park, such thickets of *A. nilotica* can reach heights up to 4.5 m and can be very difficult to penetrate. In addition, it was observed in the field that groundcover was rarely present under the *A. nilotica* canopy, which suggests possibility of an allelopathic effect (Caesariantika *et al.* 2011; Djufri 2012; Sabarno 2002; Setiabudi *et al.* 2013; Tjitrosoedirdjo *et al.* 2013). However, this assumption remains speculative and requires further testing.. Allelopathy may contribute to the capability of particular exotic species becoming dominants in invaded plant communities through the release of chemical compounds into the environment which suppress other species. In South China, the exotic plant species 'mile a minute'

(*Mikania micrantha*) released a photochemical which affected native plants and has invaded many forests in South China causing significant damage (Bao *et al.* 2009).

The capability of *A. nilotica* for long distance dispersal through herbivores (as shown in this study) and possibly also via flood water is considered a character of this species that enhances its invasive potential (Djufri 2004; Radford *et al.* 2001a). Average seed production per tree of *A. nilotica* is also high (~175,000 per tree) and these seeds can have a long life span. The parent tree itself could live for around 30 years (Djufri 2004). Lastly, it is its ability to exist in a wide range of environmental and climatic conditions that also contributes to its invasiveness. *Acacia nilotica* is also predicted under global climate change to increase its potential distribution in Indonesia (Chapter 5) and significantly increase the area at risk of invasion in Australia (Kritikos *et al.* 2003). Understanding the dynamics and seed biology characteristics of invasive alien species in native communities is important when drafting management plans for ecosystems (Booth *et al.* 2003), especially when preparing restoration programs. Therefore, the results from this study may also be relevant to other savannas, particularly other tropical savannas of South East Asia where there are often similarities in terms of community composition and herbivores.

Chapter 7: Habitat Structure of the Endemic Bali Starling (*Leucopsar rothschildi* Stresemann) in Bali Barat National Park

ABSTRACT

Tropical savannas and dry forests in Indonesia are important types of ecosystems which provide habitat to support various endemic wildlife. Several of these endemic species are now seriously threatened and accordingly have high conservation status according to the IUCN, including the Bali starling (*Leucopsar rotschildi*) which is mostly now restricted to Bali Barat National Park. Given the high extinction risk facing such species, conservation programmes are likely to require multidisciplinary approaches that address both the biological attributes of the species itself, as well as their habitat requirements. Regrettably, for many species, their habitat ecology remains inadequately understood. The objectives of this chapter are to: 1) characterise the habitat of the Bali starling in terms of vegetation structure and floristic composition; and 2) document evidence of vegetation cover changes in Bali Barat National Park, both in areas of known bali starling occurrence and generally. Analysis of remote sensing imagery as well as field sampling of vegetation attributes was conducted to address these objectives. Normalized Difference Vegetation Index (NDVI) was calculated from Landsat imagery using red and near infrared bands. Tree cover percentage data was obtained from the Vegetation Continuous Fields (VCF) product of the University of Maryland. Results showed that forest and savanna are the dominant land cover types in Bali Barat National Park, but their distribution is somewhat dynamic with changes in vegetation cover and greenness found across the years in which increasing cover of woody plants is the general trend. In Bali Barat National Park, the Bali starling is mostly found at or near distinct vegetation boundaries, such as the border between savanna-forest; savanna-cropland; savanna-shrubland; settlement-cropland; and forest-shrubland. Although the Cekik area had plant species that has been known to be able to provide shelter and food for Bali Starling (as was Brumbun), the bird has not been observed in this area since the 1990s. These results further confirm the importance of examining habitat patterns of endemic bird within a landscape that are influenced by multiple factors that interact in space and time. Addressing data shortage in habitat patterns within endemic species distribution is important for conservation managers developing conservation management strategies. Evaluating the remaining habitat of the species is important for the conservation of Bali starling and useful for the reintroduction and release program into their natural habitat.

Keywords: Tropical savanna, dry forest, habitat structure, habitat suitability, species distribution model, *Leucopsar rotschildi.*

7.1. INTRODUCTION

Tropical savannas and dry forests are important types of ecosystems which comprise habitat that supports various endemic wildlife of Indonesia. Some of these species are now under serious threat of extinction and, consequently, have high conservation status according to the IUCN (IUCN 2014), for example, Javan banteng (*Bos javanicus* subsp*. javanicus*), now mostly confined to savanna Baluran National Park in East Java, the Komodo dragon (*Varanus komodoensis*), endemic only to Komodo Islands of East Nusa Tenggara, and the endemic bird Bali starling (*Leucopsar rotschildi*), which is now mainly found in the savanna of Bali Barat National Park on the north-west tip of Bali. However, rapid and widespread habitat loss and variable capacity in natural reserve management pose considerable risk to these species and Indonesian biodiversity in general (Purwandana *et al.* 2014).

In terms of the major land uses categories, forests only cover ~128, 000 ha (23%) of the total land area of Bali. Of this area, it is estimated that approximately 31,817 ha (25%) has been degraded or more recently converted to other land uses (Hehanusa *et al.* 2005). These land use changes are caused by various factors such as fires, illegal logging, buildings/infrastructure, and other developments that are not in line with the forestry sector (Hehanusa *et al.* 2005). One of the remaining significant forested areas in Bali is the Bali Barat National Park (BBNP). Beside monsoon and tropical rain forest, the national park also includes other important ecosystems such as savanna, mangroves and near-shore coral reefs. The national park harbours more than 11 species of mammals, more than 160 species of birds and herpetofauna, and more than 120 species of coral reef fish (Wiryawan 2014). One species of bird is of particularly high conservation significance in Bali Barat National Park, the endemic the Bali starling, often also known as Bali mynah (*Leucopsar rothschildi*).

The critically endangered Bali starling is the only endemic bird found in Bali. Stresemann (1912) collected and described the first Bali starling known to science near Bubunan, Bali. The Bali starling is an attractive aviary bird being largely white, with black wing and tail tips and with bare skin of a turquoise-blue colour on the lores and behind the eye (see Figure 3 in Chapter 8). It has an erectile crest, slightly longer in males (Greenwell 1980). Bali starlings are omnivorous but feed their young an insectivorous diet (Seibels *et al.* 1997). Bali starlings are reported to eat ants, ant pupae, beetles, bark borers, caterpillars, corn grubs, crickets, dragonflies, earthworms, grasshoppers large flies, moths, small reptiles, spiders and termites (Collins and Smith 1994; Collins *et al.* 1998; van Balen and Gepak 1994). Bali starlings nest in trees 4–10 m above ground by using natural cracks or hollows in trees (Williams and Feistner 2006). Bali starling also frequently make use of old nest holes of barbets (*Psilopogon* spp.) and woodpeckers (Picidae) (Williams and Feistner 2006).

On account of its restricted range, extremely small numbers in the wild and pressures on the last free ranging birds, the Bali starling is considered critically endangered according to the latest International Union for Conservation of Nature (IUCN) threat categories (IUCN 2014). Habitat destruction and capturing for the pet trade brought the species to the verge of extinction (van Balen *et al.* 2000). It was estimated that less than 20 birds remained in the wild in 1998 within a small area on the northwest tip of the island within the boundaries of BBNP (Collins *et al.* 1998). Although the wild population is near extinction, Bali starlings have been successfully bred in captivity (Collins *et al.* 1998).

Animals choose habitat based on a diverse range of environmental factors operating across various scales, and thus knowledge regarding these factors is particularly important for species at risk of extinction. Where available, such information can be used to direct habitat restoration, to recognize priority areas for biodiversity conservation and to offer insights into likely impacts of environmental change (Howland *et al.* 2016). The original habitat of the Bali starling in Bali was described as 'dry savanna and shrub woodlands' and 'tall and dense forest' in the 1920s (van der Paardt 1926), and at this time was believed to be historically restricted to a narrow belt of dry monsoon climate in Northern Bali and East Java (van Balen *et al.* 2000). Its range shrunk considerably between 1920-1960 to the fire-induced open shrub and savanna woodland, found below an elevation of 150–175 m in the north-east part of the Prapat Agung peninsular within the BBNP (van Balen *et al.* 2000). Patterns of bird distribution and abundance within a landscape are influenced by multiple factors that interact in space and time (Orians and Wittenberger 1991). Habitat structure and floristic composition, such as percent canopy cover, tree species diversity and the distribution of specific plant taxa, are known to have a significant role in defining the occurrence of bird species in space (James and Wamer 1982; Rice *et al.* 1984; Wiens and Rotenberry 1981).

Given the high extinction risk facing such species, conservation programmes are likely to require multidisciplinary approaches that address both the biological attributes and resource requirements of the species itself, as well as their habitat requirements and conditions (Estoque *et al.* 2012). Evaluating the remaining habitat is important in conservation of Bali starling and useful for the reintroduction and release program to their natural habitat. Moreover knowledge and understanding the likely potential distribution and habitat suitability for the Bali starling has important implications for selecting potential sites for future *ex-situ* conservation and breeding programs designed to save this endemic bird from extinction.

There have been many studies on the Bali starling; however most of these have focussed on the bird itself ranging from its behaviour, reproduction, breeding, genetic, taxonomy, demography and reintroduction, among others (Collins and Smith 1994; Collins *et al.* 1998; De Iongh *et al.* 1982; Dirgayusa *et al.* 2000; Seibels *et al.* 1997; Williams and Feistner 2006). However, studies with regards to the Bali starling's habitat are scarce (but see Widodo 2014), especially considering that these habitats are vital to the ongoing maintenance of viable populations of Bali starling, but yet are also very prone to conversion, disturbances and degradation. Therefore the objectives of this chapter are to: 1) describe the known current distribution of Bali starling and characterize its habitat structure in Bali Barat National Park in terms of its plant community structure and composition, 2) assess the cover and greenness index (NDVI) to quantify the dynamics of vegetation cover in these habitat areas; and 3), assess the degree of recent habitat change affecting the species more broadly in BBNP.

7.2. MATERIALS AND METHODS

7.2.1. Study Site

Bali Barat National Park (BBNP) is located on the north-western side of Bali, Indonesia. The park covers around 19,000 ha which comprises of 15,588 ha of terrestrial areas and 3,415 ha of marine habitats. This area is approximately 5% of Bali's total land area. A seaport at Gilimanuk is located west of the park. The national park also bordered with several villages. Bali Barat National Park can be reached by roads from Gilimanuk and Singaraja, or by using ferries from Ketapang, East Java. There are several major habitat types in the national park: savanna, mangroves, montane and mixed-monsoon forests, and coral islands. Bali Barat was designated as a National Park in 1984 based on Ministry of Forestry decree (No. 096/Kpts-II/1984). Its area has mostly latosol type of soil being reddish in colour, weakly crumbed and sticky when wet although hardening and cracking when dry. The park is topographically varied ranging from plains near the coast to steep hills and mountains. There are four mountains, namely Prapat Agung, Banyuwedang, Klatakan and Sangiang *(*the highest at 1,414 m). Off the coast, there are four islands that are also under BBNP management jurisdiction, namely Menjangan, Burung, Gadung and Kalong Islands. Bali Barat National Park has a moderately seasonal climate with rain in each month, but more falling in the wet season (December to February). Annual average rainfall ranges from 900 – 1,500 mm and average temperature is 33°C (Masy'ud *et al.* 2008; Masy'ud *et al.* 2007; Whitten *et al.* 1996).

7.2.2. Bali Starling Distribution in Bali Barat National Park

A number of reports on the Bali starling population, distribution and their habitat were obtained through a review of literature and also personal communications with BBNP rangers and managers. Locations of Bali starling were obtained from De Longh *et al.,* (1982), Whitten *et al.,* (1996), van Balen *et al.*, (2000) and BBNP manager, Wiryawan, (2014, pers. comm.). Based on these data I divided Bali starling occurrence data into three eras of distribution, namely 1984, 1994 and 2010, because I considered these as the only 3 years in which reliable and accurate surveys were done and so these years were only used. I then conducted overlay analysis of these Bali starling locations data with Indonesia's topographical/earth surface map (Rupa Bumi Indonesia/RBI) for year 2001 (scale 1 : 80,000) obtained from Indonesian Geospatial Agency (BIG/BAKOSURTANAL). I used 2001 land use data because this was the most recent available. All data used the same datum and map projection within the GIS (WGS 1984-UTM, Zone_50) to avoid misalignment.

7.2.3. Habitat Structure

7.2.3.1. Vegetation Cover and Greenness Index

A) Vegetation Cover

The Vegetation Continuous Fields (VCF) collection contains proportional estimates for vegetative cover types: woody vegetation, herbaceous vegetation, and bare ground. The product is derived from all seven bands of the MODerate resolution Imaging Spectroradiometer (MODIS) sensor onboard NASA's Terra satellite. The continuous classification scheme of the VCF product may depict areas of heterogeneous land cover better than traditional discrete classification schemes. While traditional classification schemes indicate where land cover types are concentrated, this VCF product is very good at showing how much of a land cover class such as "forest" or "grassland" exists anywhere on a land surface (DiMiceli *et al.* 2011).

Estimated tree cover percentage for Bali Barat National Park was obtained from MODIS Vegetation Continuous Field product (only available for 2000 to 2013). This MODIS product is freely available from the web http://glcf.umd.edu/data/vcf/. The area including BBNP was downloaded as GeoTIFF files using the 4950 tile/path of the repository directory. The data is already geo-rectified, and WGS 1984 zone 50 was used as spatial reference. Each pixel has a specific code value that refers to specific label (Table 1).

Table 1. Code value and the corresponding label in the Vegetation Continuous Field (VCF) product.

Code Value	Label
0-100	Percent of pixel area covered by land cover type
200	Water
253	Fill (null)

This MODIS VCF tree cover percentage data from 2000 to 2010 were then overlaid with the 2010 Bali starling distribution. Changes in tree cover percentage in each location of the Bali starling 2010 distribution was then extracted by identifying the mean value of the pixels values at the exact coordinate and on the nine pixels immediately surrounding the coordinate. The changes in mean VCF tree cover (calculated across all locations of the species) across different years were tested for significance using ANOVA in SPSS and when significant difference detected, a post-hoc test was then performed.

B) Greenness Index

NDVI is an index derived from remotely sensed imagery which can differentiate between vegetation types by showing the difference between near infrared (which is strongly reflected by vegetation) and red light (which is absorbed by vegetation). NDVI is correlated to vegetation biomass, vigour and photosynthetic activity. This index exploits the reflectance patterns of ground elements in the red (R) and near-infrared (NIR) bands of the electromagnetic spectrum to distinguish green vegetation from its background soil brightness, and is calculated as (NIR - R)/ (NIR + R). NDVI values range from -1 to 1, with positive values representing vegetated areas and negative values representing non-vegetated regions (Sankaran 2001). The NDVI ratio approach usually adopted for land cover change estimation is used here in preference to the more commonly employed post-classification pixel-by-pixel comparison method (Lillesand *et al.* 2008) since it also permits identification of areas where changes in vegetative cover have been significant, but insufficient to cause change in class membership (Sankaran 2001).

In order to generate normalized difference vegetation index (NDVI), a number of Landsat images were used. Landsat images were downloaded from http://earthexplorer.usgs.gov/ path 117, row 066. When selecting images to be download, I looked for images with minimal cloud cover percentage by selected scenes with at least image quality level 9 (no errors detected, perfect scene). Details of each image downloaded can be seen in Table 2.

Table 2. Details of images downloaded for NDVI analysis.

Images	Source	Date acquired	Spatial Resolution	Image quality	Cloud cover
1	Landsat 4	21/03/1989	30 x 30 m	9	20
2	Landsat 7	12/11/1999	30 x 30 m	9	8.63
3	Landsat 7	31/05/2003	30 x 30 m	9	7.53
4	Landsat 8	11/06/2016	30 x 30 m	9	24.99

NDVI was generated using the NDVI feature in ArcMAP (Arc GIS 10.1) image analysis toolbar. Band 1, 2, 3, and 4 were chosen for Landsat 4 and 7, whereas band 2, 3, 4, and 5 were chosen for Landsat 8 as input images in ArcMAP which represent the blue, green, red and near infra red (NIR) bands. By choosing image analysis tab, all the bands layers were merged into one composite layer and then the RGB channels were adjusted to just the NIR, red and green bands to extract the NDVI values. Once NDVI images were generated, different levels of a green colour scheme was applied for easier interpretation. I kept in mind that these NDVI values represent just one time in the year (due to the limited availability of good images for the chosen years) so it was used carefully as NDVI will mainly reflect recent rainfall especially in terms of groundcover. Then, data points for the 1984, 1994 and 2010 Bali starling locations were overlaid on the NDVI images from different years (1989, 1999, 2003 and 2016). These years were chosen because these years were the closest years to the years of Bali Starling location (as clear image of the exact years of Bali Starling location could not be found). The mean and SD from nine pixels values surrounding the exact coordinate locations of the Bali starling was then calculated from the NDVI images. Changes in mean NDVI of starling locations between different years were tested for significance using ANOVA in SPSS and, when significant difference detected, a post-hoc test was then performed.

7.2.3.2. Plant Community Structure and Composition

Brumbun was a site where Bali starling was regularly observed in the period of 1979 to 1994. There was no surveys done in this area during 1995 – 2009, therefore no data on Bali starling in these locations could be obtained. Recent surveys were conducted by Bali Barat National Park in 2010 and found Bali starling in this location again. At another site, Cekik, the Bali starling was also observed in the period 1979-1994, but in the recent survey in 2010 by the BBNP management it was not found again in the area. Fieldwork was conducted in November 2014 in these two locations, namely Cekik and Brumbun. The study areas were cross-checked with a fire map based on MODIS burn area product from year 2000 to 2013 to obtain information on fire history (which showed no fires at these localities). Additionally by examining the Landsat imagery there appears to be no major fires in BBNP over this period.

In September to November 2014 (dry season), ten sampling plots (50 x 50 m) were established randomly in each savanna sites (Cekik, Brumbun). In each of the 50 x 50 m plots, I nested smaller plots of 5 x 5 m randomly. Inside the 50 x 50 m plot I identified, measured and recorded all tree species ≥ 10 cm diameter at 1.3.m (dbh). In the smaller-nested plots I noted all groundcovers species (grasses, herbaceous and ferns) and estimated their cover. Species in each site were identified in the field where possible and a field herbarium created so that the plants can be more easily recognised to species level in the subsequent field work, which demands rapid identification in the field. The identification was assisted by herbarium Bogoriense and herbarium Baliensis of the Indonesian Institute of Sciences (LIPI). Plant identification made use of flora books such as "Flora of Java" (Backer and van den Brink 1963), "Mountain Flora of Java" (van Steenis 1972), "Weeds of Rice in Indonesia" (Soerjani *et al.* 1986), " Ecology of Java and Bali" (Whitten *et al.* 1996) and "Ecology of Nusa Tenggara and Maluku" (Monk *et al.* 2000) and names standardised based using the Plant List (www.theplantlist.org).

Importance Value Index or IVI (Kent 2011; Kent and Coker 1992) was calculated for each species in each plot to understand the structure and plant community composition in each of the savanna. Importance Value Index (IVI) (Curtis and McIntosh 1950; Kent and Coker 1992) was used to describe the quantitative structure of the community. This statistic represents the contribution that a species makes to the community in terms of the number of plants within the quadrats (density), its contribution to the community through its distribution (frequency), and its influence on the other species through its dominance. Importance Value Index was calculated for each species of tree and groundcover in each of the study sites. The formula for tree IVI is as follow:

$$IVI = RD + RF + RDom$$

Where RD = relative density of a species, RF = relative frequency of a species and RDom = relative dominance of a species.

$$\text{Relative Density of species A} = \frac{\text{Number of individual of A species}}{\text{Total number individual of all species}} \times 100\%$$

$$\text{Relative Frequency of species A} = \frac{\text{Frequency value of A species}}{\text{Total frequency value of all species}} \times 100\%$$

$$\text{Relative Dominance of species A} = \frac{\text{Dominance value of A species}}{\text{Total dominance value of all species}} \times 100\%$$

Dominance values for a tree species were obtained by dividing the basal area of the tree with the size of the plot (Simon 1996; Supriyadi and Marsono 2001). The IVI formula for groundcover species (including seedlings) was similar to the tree layer but without the calculation of relative dominance (Kusmana 1995), and so the formula is as follow:

$$IVI = RD + RF$$

Where RD = relative density of a species, and RF = is relative frequency of a species.

In addition to IVI, Shannon – Wiener species diversity and species richness was also calculated for each plot and savanna. Changes in these values tested for significance using ANOVA and post hoc Tukeys'[b] test in SPSS.

Using abundance data (cover), I tested for differences in plant community composition between Brumbun and Cekik savanna. The data were square-root transformed prior to constructing a resemblance matrix based on Bray-Curtis similarity (Valessini 2009). A Non-metric Multidimensional Scaling (NMDS) ordination diagram was then generated based on the resemblance matrix. The difference in species composition between savannas was then tested for significance using one-way ANOSIM (analysis of similarity). The R_{ANOSIM} statistic values, generated by ANOSIM, are a relative measure of separation of the *a priori* defined groups. A zero (0) indicates that there is no difference among groups, while one (1) indicates that all samples within groups are more similar to one another than any samples from different groups (Clarke 1993). This multivariate analyses made use of the PRIMER v.6 package (Clarke and Gorley 2005).

7.3. RESULTS

7.3.1. Bali Starling Distribution in Bali Barat National Park

In Bali Barat National Park, Bali starling has been mostly found in or near vegetation boundaries. It has been found at or near the border between savanna-forest, savanna-cropland, savanna-shrubland, settlement-cropland, and forest-shrubland (Figure 1). Ten out of the 21 records are within close proximity of a major vegetation boundary. Bali starling locations fall within the following land-use categories, namely: primary forest, savanna, shrubs (areas dominated by shrubs, also known as shrubland), mangrove and cropland (however it should be kept in mind that the land use map is from 2001 only).

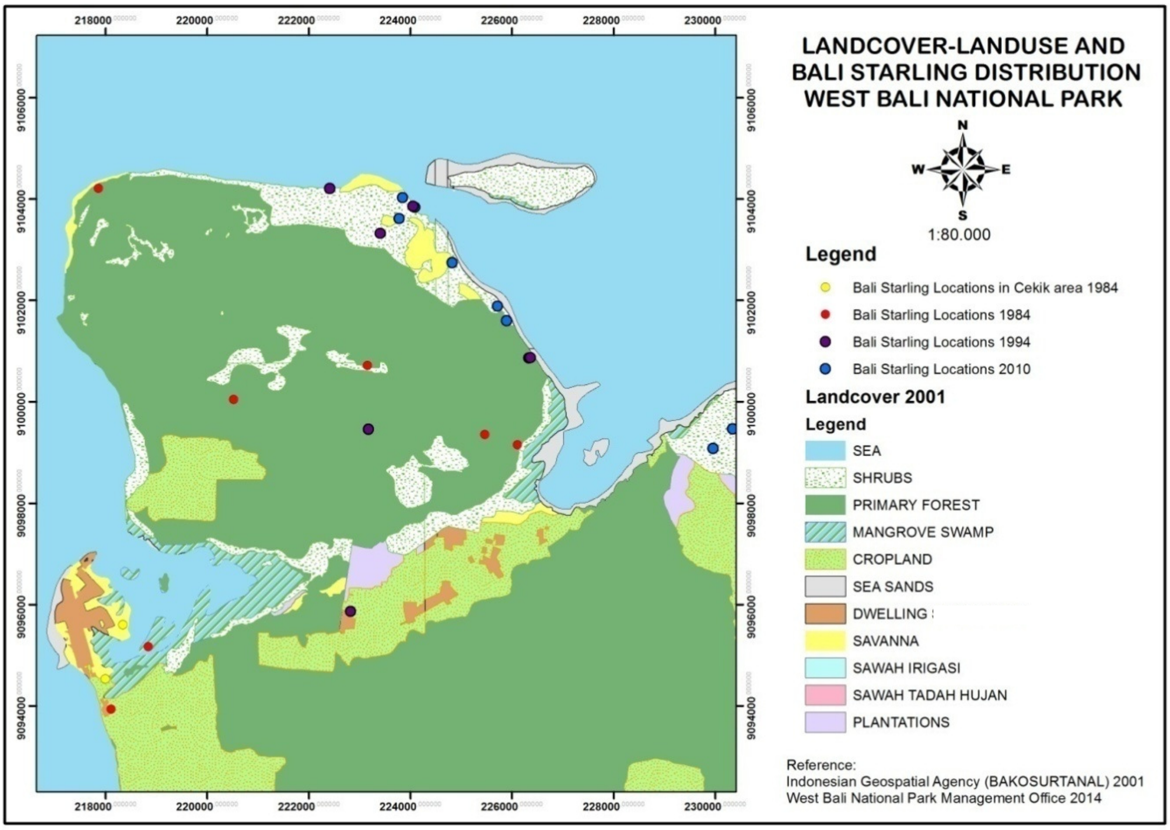


Figure 1. Overlay of Bali Starling occurrences with the 2001 Land Use map of Bali Barat National Park. 'Sawah irigasi' refers to irrigated paddy field. 'Sawah tadah hujan' refers to rain-fed paddy field.

7.3.2. Habitat Structure

7.3.2.1. Vegetation Cover and Greenness Index

In general, there was noticeable variation in the VCF tree cover percentage values from 2000 to 2010 in Bali Barat National Park (Figure 2), but areas with low tree cover have noticeable decreased in size. In most of the 2010 Bali starling locations, the VCF tree cover values have increased over the previous decade (Figure 3). Overall mean tree cover percentage also shows significant (<0.1) increasing trend during the period of eleven years (2000 to 2010) although there was a clear reduction in cover in 2007 across all sites (Figure 4).

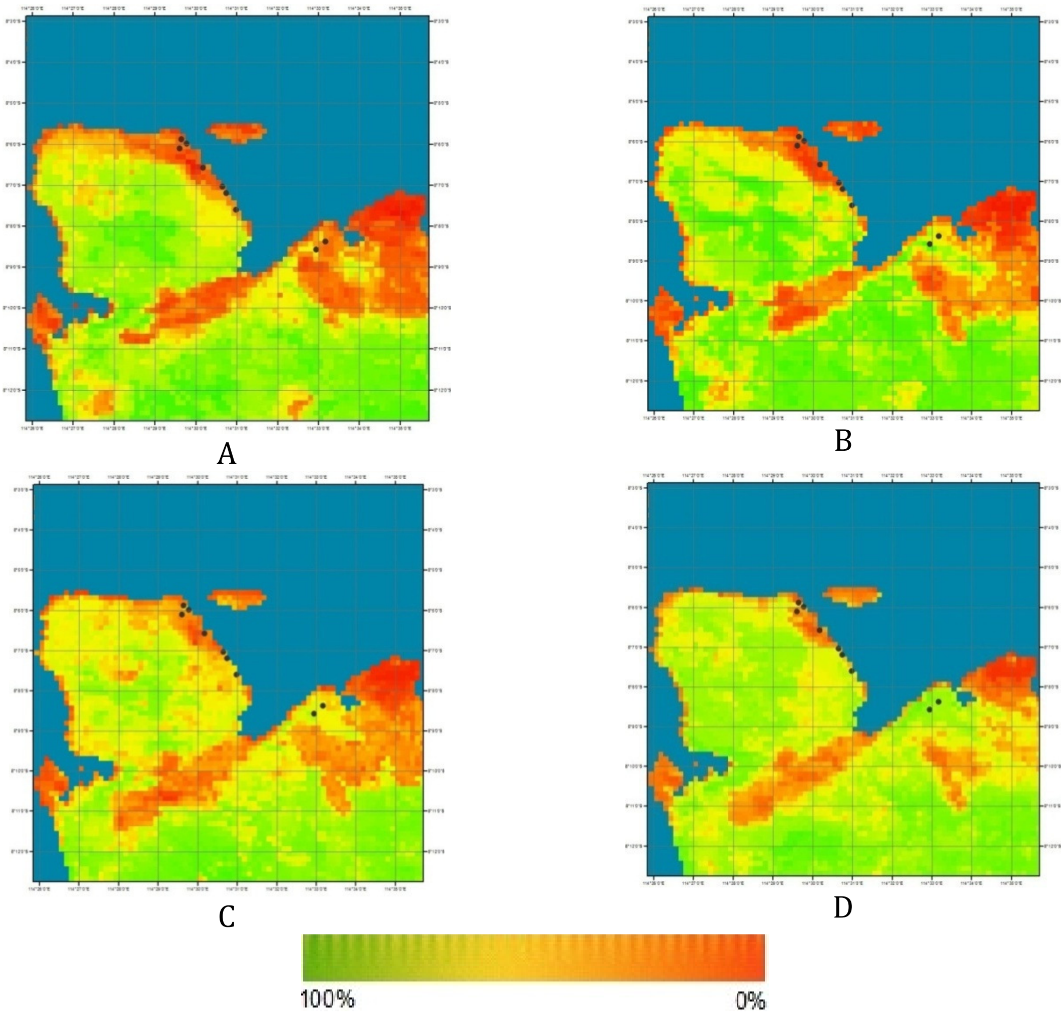


Figure 2. Maps showing changes in VCF tree cover percentage value over time from 2000 (A), 2002 (B), 2007 (C) and 2010 (D) in Bali Barat National Park. Also shown is the distribution of Bali starling in 2010 (shown as black dots).

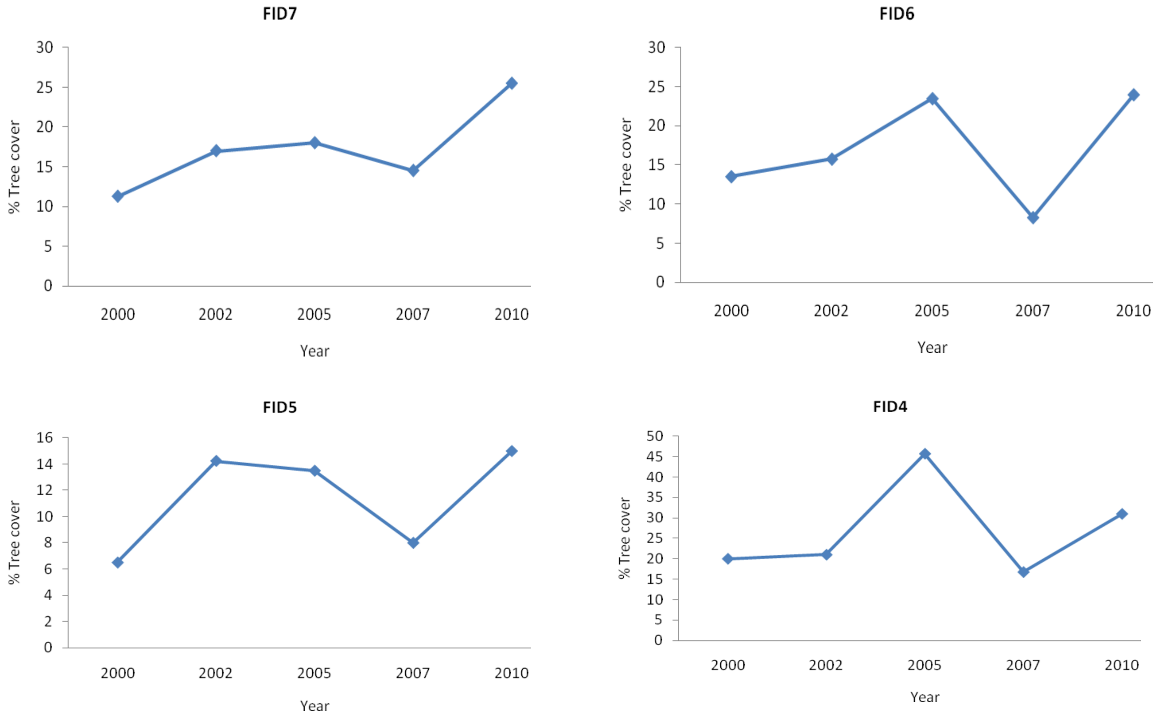


Figure 3. Changes in tree cover percentage (vertical axes) over time of each location where Bali starling was spotted in 2010 in Bali Barat National Park. Note: FID 4,5,6,7 are name of the points location on the map.

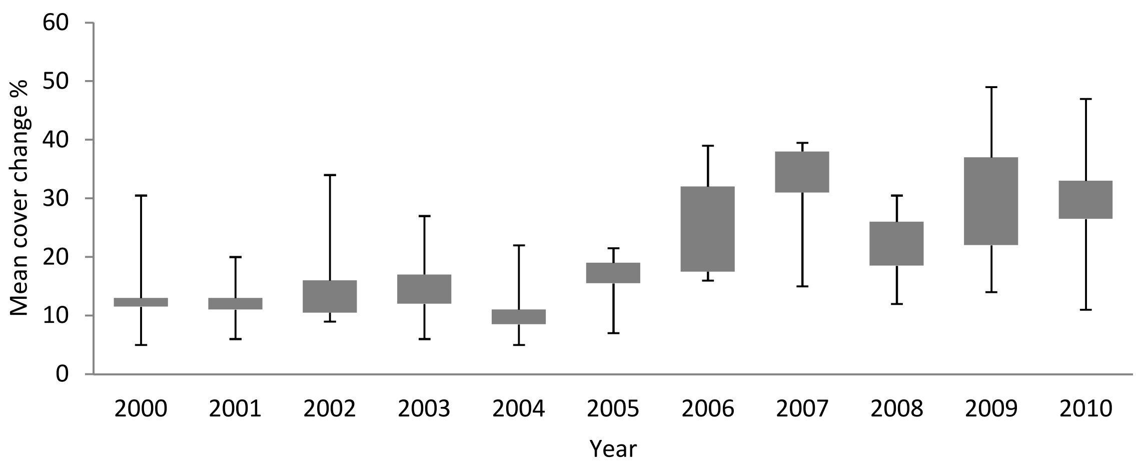


Figure 4. Box plot of tree cover changes at Bali starling 2010 sightings sites (2000 to 2010) in Bali Barat National Park.

Based on NDVI image analysis, there is a shift in habitat in the distribution of the Bali starling, where the species tends to opt out from the primary forest type of habitat and moved to a less dense, more open vegetation areas such as the dry forest/monsoon forest, secondary forest, and also savanna and grassland, over time (Figure 5).

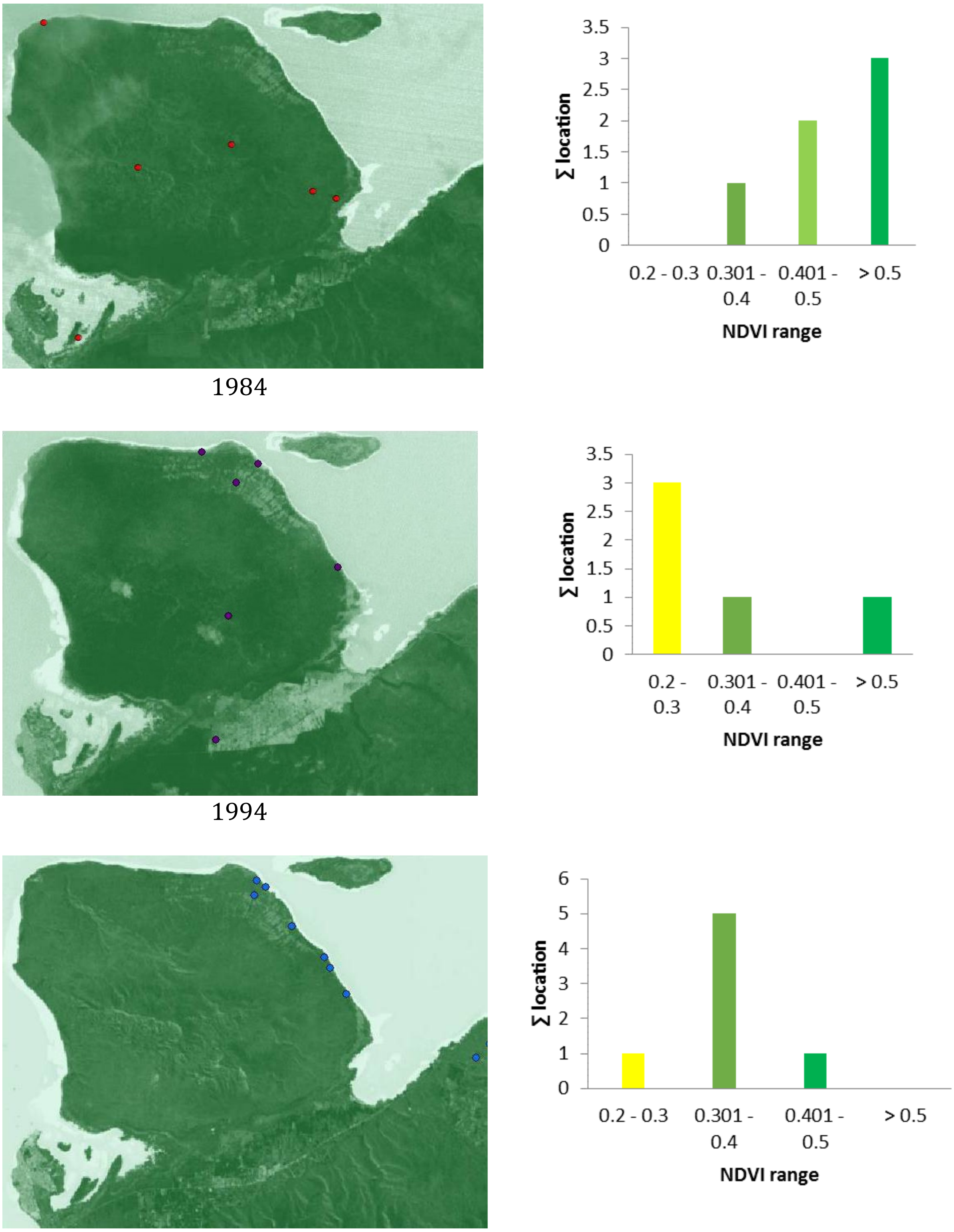


Figure 5. Bali starling habitat distribution based on their Normalized Difference Vegetation Index (NDVI) range class. Habitat and localties are presented for 1984 (upper, alligned with NDVI image date 21/03/1989), 1994 (middle, alligned with NDVI image date 12/11/1999) and 2010 (lower, alligned with NDVI image date 11/06/2016). Ranges of NDVI: 0.2-0.3 = Cropland-grassland-savanna; 0.301-0.4 = Savanna-shrubland-mangrove; 0.401 – 0.5 = Dry forest-monsoon forest; > 0.5 = Primary Forest-broad leaves-evergreen forest. NDVI range classes follow Siswoyo (2014).

7.3.2.2. Plant Community Structure and Composition

I discovered as many as 22 plant species belonging to 14 families in the two savannas. At Cekik there were 10 species and eight families, whereas at Brumbun there was 20 species within 12 families. There were significant differences ($R_{ANOSIM} = 0.228$; $P<0.003$) in Bray-Curtis species similarity between the savanna sites (Figure 6).

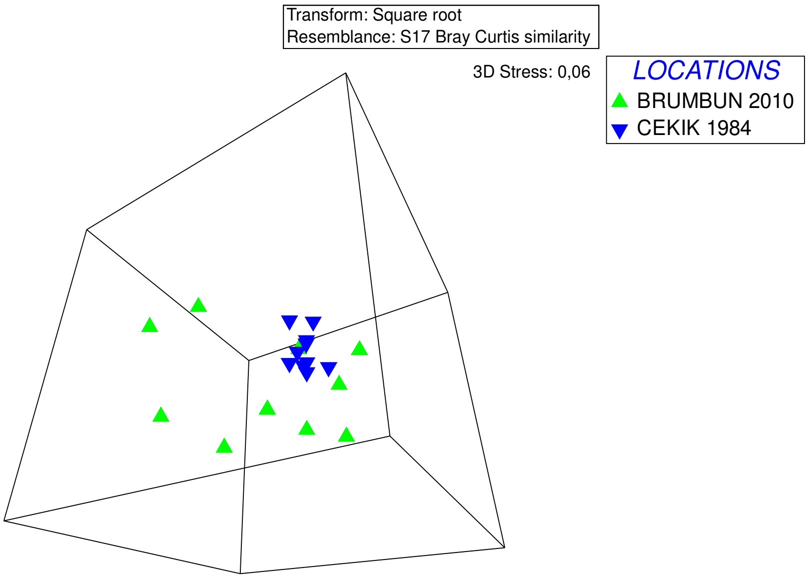


Figure 6. Non metric Multi Dimensional Scaling (NMDS) ordination based on Bray Curtis Similarity index on plant species abundance and composition between Brumbun and Cekik savanna areas in Bali Barat National Park.

There was eight species that were present in both savannas (Cekik and Brumbun) namely *Chromolaena odorata, Lantana camara, Desmodium laxiflorum, Grewia eriocarpa, Bridelia stipularis, Cynodon dactylon, Calamagrostis australis,* and *Ziziphus mauritiana* (synonym *Z. jujuba*). There was no significant difference ($P>0.05$) in Shannon-Wiener species diversity between Brumbun and Cekik, however species richness among the two savannas was significantly different ($P<0.05$; Figure 7), with Brumbun having more species compared to Cekik.

Based on cover values (IVI) of tree habit in Cekik, *B. stipularis* was the most dominant species with IVI of 73.75 and relative density of 43.75. Second next to *B. stipularis* was *Borassus flabellifer* with IVI of 61.25 and relative density of 31.25 (Table

5). Other species of trees that were also presences in Cekik were *Z. mauritiana* (IVI 32.5), *G. eriocarpa* (IVI 16.25) and *Schleicera oleosa* (IVI 16.25). Groundcover layer in Cekik is heavily dominated by native grass *C. australis* (112.67) and with considerable amount of invasive exotic herbaceous such as *C. odorata* (47.19) and *L. camara* (26.14) (Table 5).

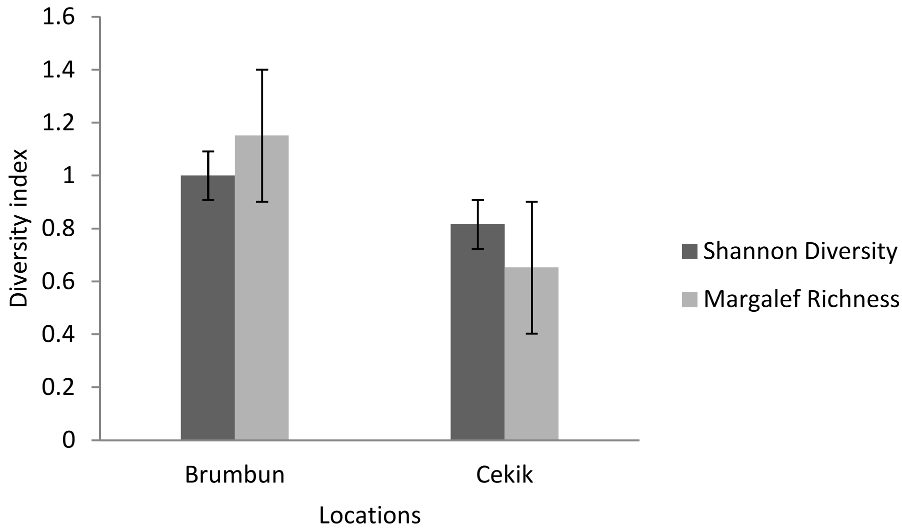


Figure 7. Comparison of Shannon-Wiener species diversity and Margalef's species richness in Brumbun and Cekik savanna, Bali Barat National Park.

The tree layer in Brumbun is dominated by *Z. mauritiana* and *Glochidion sumatranum* (32.05), followed by *Phylanthus emblica* (23.71) (Table 5). The composition of the groundcover layer is generally similar to Cekik, but with different proportions. The native grass species *C. australis* dominates the site (IVI 53.92), followed by invasive exotic forb *C. odorata* (IVI 32.62) and also other grass species such as *C. dactylon* (IVI 24.43) (Table 3). Tree cover and IVI is higher overall at Cekik compared to Brumbun.

Table 3. Importance Value Index of plant species in each of the savannas in Cekik (Bali starling distribution area up to 1994) and Brumbun (Bali starling distribution area in 1994 and 2010) Bali Barat National Park.

Species	Family	Relative Density	IVI
	Cekik		
Trees			
*Bridelia stipularis***	Phyllanthaceae	43.75	73.75
*Borassus flabellifer***	Arecaceae	31.25	61.25
Ziziphus mauritiana (syn *Z. jujuba*)**	Rhamnaceae	12.5	32.5
*Grewia eriocarpa***	Malvaceae	6.25	16.25
Schleichera oleosa	Sapindaceae	6.25	16.25
Groundcover			
Calamagrostis australis **	Poaceae	79.96	112.67
*Chromolaena odorata**	Asteraceae	15.05	47.19
*Lantana camara**	Verbenaceae	4.71	26.14
Cynodon dactylon	Poaceae	2.61	6.18
*Desmodium laxiflorum***	Fabaceae	0.52	4.09
*Borassus flabellifer****	Arecaceae	0.13	3.7
	Brumbun		
Trees			
Ziziphus mauritiana (syn *Z. jujuba*)**	Rhamnaceae	15.38	32.05
*Glochidion sumatranum***	Phyllanthaceae	15.38	32.05
Phyllanthus emblica	Phyllanthaceae	15	23.71
*Flacourtia rukam***	Salicaceae	7.69	16
*Streblus asper***	Moraceae	7.69	16
Azadirachta indica	Meliaceae	7.69	16
*Grewia eriocarpa***	Malvaceae	7.69	16
Acacia leucophloea (syn *Vachellia leucophloea*)**	Fabaceae	7.69	16
*Bridelia stipularis***	Phyllanthaceae	7.69	16
*Alstonia spectabilis***	Apocynaceae	7.69	16
Groundcover			
*Calamagrostis australis***	Poaceae	42	53.92
*Chromolaena odorata**	Asteraceae	15.95	32.62
Cynodon dactylon	Poaceae	17.28	24.43
Glochidion sumatranum ***	Phyllanthaceae	6.91	14.05
*Lantana camara**	Verbenaceae	2.65	9.8
Flacourtia rukam ***	Salicaceae	4.52	9.28
*Desmodium laxiflorum***	Fabaceae	3.19	7.95
*Merope angulata***	Rutaceae	1.86	6.62
*Micromelum minutum***	Rutaceae	0.79	5.55
Saccharum spontaneum	Poaceae	1.06	3.44
*Alstonia spectabilis****	Apocynaceae	0.53	2.91
*Grewia eriocarpa***	Malvaceae	0.53	2.91

*Ziziphus mauritiana ****	Rhamnaceae	0.26	2.64
*Croton tiglium***	Euphorbiaceae	0.26	2.64
*Indigofera tinctoria***	Fabaceae	0.26	2.64
*Streblus asper***	Moraceae	0.26	2.64
Azadirachta indica	Meliaceae	0.26	2.64

*Refers to Invasive Alien Species (IAS). ** Refers to native (Southeast Asia). ***Indicates seedling form. Source of information: GBIF (Global Biodiversity Information Facility), ALA (Atlas of Living Australia), Flora Malesiana, PROSEA, Catalogue of Indonesian Botanical Garden.

Plant species composition was categorized into different uses by the Bali starling. The usage categories empolyed were: food, shelter (nesting) or a combination of both (Table 4). In terms of plants which provide food source, Cekik generally has higher cover. Six species were categorized as food source for Bali starling namely *Z. mauritiana, G. eriocarpa, S. oleosa, Streblus asper, Azadirachta indica,* (tree species) and *L. camara* (herbaceous). Two species were used for shelter namely *B. flabellifer* (Arecaceae) and *Acacia leucophloea* (synonym *Vachellia leucophloea*) (Fabaceae). The latter (*A. leucophloea*) was also utilized by Bali starling as food source (combined). There were used plant species that were only presentin Cekik only and not in Brumbun (*S. oleosa* and *B. flabellifer*), as well as in Brumbun only and not Cekik (*S. asper, A. indica* and *A. leucophloea*). There were also plant species that were present in both of the locations such as *Z. mauritiana, G. eriocarpa* (tree species) and the invasive exotic climber *L. camara* (Table 4).

Table 4. Plant species used by Bali starling in sampling sites (Brumbun and Cekik) Bali Barat National Park

Species	Family	Habitus	Usage	Found at and IVI
Ziziphus mauritiana	Rhamnaceae	Tree	Food	Cekik (32.5) and Brumbun (32.5)
Grewia eriocarpa	Malvaceae	Tree	Food	Cekik (16.25) and Brumbun (16)
Schleicera oleosa	Fabaceae	Tree	Food	Cekik (16.25)
Borassus flabellifer	Arecaceae	Tree	Nest	Cekik (61.25)

Streblus asper	Moraceae	Tree	Food	Brumbun (16)
Azadirachta indica	Meliaceae	Tree	Food	Brumbun (16)
Acacia leucophloea	Fabaceae	Tree	Nest,food	Brumbun (16)
Lantana camara	Asteraceae	Herb	Food	Cekik (26.14) and Brumbun (9.8)

7.4. DISCUSSION

The Bali starling *Leucopsar rothschildi* is critically endangered and the only endemic bird found on the island of Bali. Based on my analysis of bird observations, it is now mostly found in a relatively open vegetation (such as savanna and open shrubland) and their boundaries with other vegetation types in Bali Barat National Park, and this confirms with reports in the literature (De Iongh *et al.* 1982; Dirgayusa *et al.* 2000). The bird has been observed to be present at or near the ecotones between savanna-forest, savanna-cropland, savanna-shrub land, settlement-cropland, and forest-shrubland. This is supported by the NDVI analysis for the three years when the species localities were recorded (1984, 1994 and 2003) which also showed there has been a shift in preferred habitat of the Bali starling from primary forests in the 1980s to more open vegetation areas including, but not limited to, dry forest/monsoon forest, secondary forest and savanna. These results further confirmed the results of van Balen *et al.* (2000) who found the Bali Starling range on Bali Island has shrunk to the fire-induced open shrub and savanna woodland below elevations of 150–175 m in the north-east part of Prapat Agung peninsular of Bali Barat National Park. However, in recent years, there was also apparent shift in vegetation cover preferred by the starling, to first very open vegetation such as savanna and then medium dense vegetation such as dry forest. This observation of recent shifts is supported by VCF data which suggest that more open vegetation, such as savanna has diminished in recent years. Savanna dynamics may have a role for Bali Starling.

The apparent savanna dynamics and shift in Bali starling habitat preference, therefore, is likely to have been influenced by several interacting factors. The first possible contributing cause is the lack of recent fires in BBNP. In chapter 4, I have discussed the importance of fire as a driving factor that decides the transition between two ecosystems. One of the key findings in chapter 4 was the change from savanna or more open vegetation to dry forest without fire (or vice versa). There is apparent increase in woody cover in Bali starling habitat as shown in VCF data for where Bali starling is located in more recent times. This is likely due to lack of fire as there were no large fires in BBNP over last 20 or so years. The second possible reason is increased human population and associated infrastructure and land use (dwellings, roads, croplands) in the areas adjacent to BBNP, thereby decreasing available Bali starling habitat. This is especially true of the Cekik - Gilimanuk area which has seen a major increase in human-dominated land uses, mainly through conversion of savanna. Accounts of local people indicate that the conversion of monsoon forest to agricultural land had a negative impact on Bali starlings (van Balen *et al.* 2000). Lastly, the shift in preferred habitat is perhaps due to changes in plant species composition and vegetation structure which affect the Bali starling food sources and utilization of plants for foods and nesting.

Results of this study also noted the absence of Bali starling in the Cekik area since the mid 1990s, although this area has plant species that are known to provide shelter and food for Bali starling (e.g. *Schleicera oleosa* and *Borassus flabellifer*) (Widodo 2014) similar to other areas known to be have been successfully recolonised by Bali starling in recent times, e.g. Brumbun with its *Acacia leucophloea* (Figure 8). The reason for this might be related to several factors such as increases in human habitation that may lead to decrease in plant species richness-diversity, as well as increasing the risk of poaching. Although Brumbun is richer in plant species compare to Cekik, different species of tree dominate at these sites, although groundcover species composition between the two locations was relatively similar with exotic invasive species such as *Chromolaena odorata* and *Lantana camara* dominating the understorey. van Balen *et al.* (2000) wrote that Bali starling habitat in the late 1990s was open woodlands which were dominated by *A. leucophloea* trees with an undergrowth of *L. camara* and *C. odorata* shrubs, and *Imperata cylindrica* grass, and intersected by moister and more densely forested valleys with dominant trees *Grewia eriocarpa*, *Vitex pubescens*, *B. flabellifer* and *Schoutenia ovata.* This vegetation type might, however, be sub-optimal habitat for the Bali starling and it may have been driven there by poaching pressure (van Balen *et al.* 2000). In West Java, Fardila and Sjarmidi (2012) found that relatively low bird species diversity on the southern Bandung (urbanized areas) was attributable to humans. In other studies, Palomino and Carrascal (2006) found that bird species richness was significantly higher in natural than urban habitats in Spain.

Collins and Smith (1994) suggest that one of the factor in the decline of the Bali starling have been the conversion of savanna and forests to non-native tree plantations, crop land and villages. There is clear evidence of this in the Cekik area. The absence of Bali starling in Cekik may also due to land use changes (fragmented landscape) and increase in human presence in the area. Cekik is located near Gilimanuk, a busy port of Bali that connects the island with Java Island. Whereas Brumbun, is located in the Prapat Agung Peninsula, a more remote area of the national park and also located near the ranger's outpost on the northern tip of the national park area. In fragmented landscapes, species persistence depends on their ability to use different habitats, so that less suitable habitats may still favour the connectivity of the most suitable habitats in the landscape (Calviño-Cancela *et al.* 2012). Remaining fragments of native habitats such as forests are often surrounded by a matrix of modified semi-natural habitats, such as tree plantations and crops, which can still provide habitat for species associated to natural forests (Lindenmayer and Hobbs 2004). In Spain, fragmented land, eucalypt plantations has lower species diversity compare to native forest, but eucalypt plantations provide habitat for species typical of shrublands when young but do not contribute significantly to the maintenance of the understory biodiversity associated with native forests (Calviño-Cancela *et al.* 2012). In the case of Cekik and Brumbun, both locations provide suitable habitat for Bali starlings, although Cekik is decreasing in savanna-forest size, is more open and more human-populated. Most of the former habitat of the Bali starling has been converted into coconut and kapok plantations and human settlements. The availability of suitable habitat for the species at Cekik would be diminished in line with lack of remaining natural vegetation. The relatively high species richness at Brumbun and the distribution of specific plant taxa that influence the availability of shelter, feeding and breeding resources (*Ziziphus mauritiana, Grewia eriocarpa, Schleicera oleosa,* and *Borassus flabellifer*) contribute to the importance of this area for Bali starlings.

Poaching is the major threat to Bali starlings. In the 1960s, the Bali starling was trapped intensively to provide the demands of Indonesian, American and European private aviculturists. In 1966, the IUCN listed the species as endangered (Collins and Smith 1994). The Indonesian government responded with a 1971 law prohibiting hunting, capture and export of the bird (van Balen *et al.* 2000). However there are reports that poaching has continued in spite of efforts to increase patrols by the park rangers (Collins and Smith 1994; Dirgayusa *et al.* 2000). In Bali Barat National Park, specifically in Cekik, there is evidence of this as local rangers and I have encountered illegal poachers checking the handcrafted "pigeon holes" made by the Bali Barat Management to facilitate the survival of post-captivity bred Bali starling recently released in the area. Therefore in Cekik, I suggest favouring more protection and, where feasible, restorations of native species to best improve conservation outcomes for the Bali starling.

Figure 8. *Acacia leucophloea* savanna that has been restored from invasive alien species (IAS) domination through fire 'clearing' in Bali Barat National Park (BBNP). The savanna creates suitable habitat for Bali starling. *Acacia leucophloea* trees are used by the bird for nesting. BBNP management put handcrafted "pigeon holes" in large *A. leucophloea* trees across this savanna to create Bali starling nesting holes.

In summary, this study characterising the habitat structure, dynamics and projection of the endangered endemic Bali starling suggests that both forest and savanna are important land cover types and habitat for Bali Starling in Bali Barat National Park with changes in the relative proportions of these types, as well as availability of ecotones between them, are likely to be particularly important for this species. The increase in woody plant cover in the remaining savannas of northern BBNP where the Bali Starling are known to currently occur, and which may reflect a lack of burning in this area, is therefore of concern.The results of this study also further confirmed the importance of examining habitat characteristics and dynamics of endemic birds within landscapes that are influenced by multiple factors that interact in space and time (Fardila and Sjarmidi 2012; Orians and Wittenberger 1991). Habitat structure and floristic composition, such as percent canopy cover, tree species diversity and the distribution of specific plant taxa, are known to have a significant role in defining the occurrence of bird species in space (James and Wamer 1982; Rice *et al.* 1984; Wiens and Rotenberry 1981).

Chapter 8: General Discussion and Conclusion

8.1. INTRODUCTION

Relative to Africa, Australia and South America, little has been written on Asian savannas (Ratnam *et al.* 2016). The research presented in this study fills a gap in our understanding of an important, though understudied, Indonesian savanna ecosystems. The aims of this study were to determine the floristic patterns of the tropical savanna ecosystem, especially in the wetter regions of Indonesia, and what factors have shaped the community and its dynamics, including fire, invasive alien plant species and herbivory. In addition, I also explored the application of various remote sensing information, such as the use of Moderate-resolution Imaging Spectrometer (MODIS) and Vegetation Continuous Field (VCF) data, as well as calculation of Normalized Difference Vegetation Index (NDVI) and supervised classification of Landsat images, to examine the distribution and dynamics of savanna. The major findings of the research are identified in the next section of this chapter, and then the major themes which cut across the chapters are explored in following sections. The detailed results and discussions on each topic are presented within the relevant chapters.

In general, results from this study have shown that savanna plant community in Indonesia is formed and maintained by interaction between climatic factors, fire regime and grazing. Invasive species were also present in the studied savannas such as *Chromolaena odorata* and *Lantana camara*. These invasive species together with forest pioneer/edge specialist species (such as *Ficus septica, Laportea stimulans, Melastoma polyanthum, Nauclea orientalis, Rubus rosifolius*), may also be increasing in the absence of fire, which may, in turn, be leading to a change in state from savanna to dense woody vegetation. Absence of fire seems to be changing structure and floristics of savanna vegetation which has important implications for savanna species, including rare fauna such as Jalak Bali/Bali Starling (*Leucopsar rotschildi*) and Javan banteng (*Bos javanicus* subsp. *javanicus*). The other important invasive alien species is *Acacia nilotica* which was different from the other invasive species in that it is promoted by herbivory, and maybe also by fire. Current and future climate change is predicted to increase areas where invasion by *A. nilotica* is likely to occur especially in the eastern parts of the archipelago.

8.2. SUMMARY OF RESEARCH FINDINGS

Chapter 2 documents the characteristics and variation in the structure, diversity and composition of savanna plant communities in the higher rainfall parts of Indonesia and how they relate to environmental factors. This study compared the savanna plant community in Baluran and Alas Purwo National Park (Java Island), Bali Barat National Park (Bali Island) and Rinjani National Park (Lombok Island). The plant communities were found to be highly variable (from each other and within each other), with as many as 43 plant species within 26 families across the four savannas including one fern, seven grass or grass-like plants and two forbs (a full list of species, habit and their typical habitats are presented in Appendix 1). Each savanna has structural characteristics and dominant species that differentiated it from the others. Rinjani has the highest species diversity compared to the others, while Alas Purwo holds the lowest species diversity. Bali Barat retains the highest species richness compared to others and Alas Purwo remains the lowest in terms of species richness and species diversity as well. The main environmental variables that were found to affect the plant community composition were elevation, along with its associated climatic factors (precipitation, temperature), and then fire regime.

Chapter 3 examined spatial and temporal patterns of fires in tropical savanna and other biomes and land uses across Indonesia. The results presented in this chapter revealed that across Indonesia, fire hotspots were detected the most in year 2002 and were least in 2010. The extent of burning was mostly associated with annual Southern Oscillation Index, where most burning occurring when the SOI was less than zero. Sustained negative SOI values generally mean drier conditions across SE Asia. In Indonesia, most of the fires occurred in the middle of the austral dry season (August-September). Fires were also more obvious in the drier parts of the Indonesian Archipelago, with most burning occurring in East Nusa Tenggara (NTT) Province where Sumba and Timor Islands were the prominent ones burning. On Sumba Island, the distinct dry season lasts for at least four months, and savannas/open vegetation are the most common feature of the island which was burnt the most out of any island studied. The overall burnt areas were slightly higher in forests than sparseland (savannas and other open vegetation). However there are far more forests than savannas, so fires tend to burn open vegetation in Indonesia.

In Chapter 4, I explored the structure and composition of seasonally dry tropical forest *(*SDTF) and compared this to savanna at Baluran National Park, East Java, Indonesia to evaluate the possible evidence of transitions between SDTF and savanna. The results from Chapter 4 showed that SDTF seems to have a fairly complete structure, less diverse species and a prominent vegetative response to fire. Seasonally dry tropical forest and savanna seems to be closely related. Long unburnt savanna tends to shift towards dry forest in species composition, whereas burnt SDTF develops closer floristic similarity with savanna; therefore fire may have play a role in this savanna-forest biome shifting. Other evidence for such shifting is based on: 1) the existence of distinct borders or boundaries between savannas and adjacent forests that have no correlation with changes in soil types; 2) the occurrence of forest tree species in savannas and the swift re-growth of shrubs and trees species when fire and grazing is excluded; and 3) the history/records of intentional burning and/or natural fires in the national park (based on MODIS fire products and information from local rangers). Fire history shows that perhaps regular fires are needed to maintain the main savanna areas of the Baluran National Park.

Chapter 5 and 6 placed the present issue of invasion in the context of conversion of savanna ecosystems to a homogenous stand of prickly Acacia (*Acacia nilotica*). Results from remote sensing analysis in chapter 5 indicates that invasion of *A. nilotica* into tropical savanna is occurring rapidly in Baluran National Park. Results also show that global climate change is likely to increase the potential distribution of *A. nilotica* in Indonesia and significantly increase the area at highest risk of invasion whereby, by the year 2045, it is most likely to spread to eastern parts of Indonesia. Chapter 6 demonstrated that fire and grazing may play an important role in the invasion. The absence of repeated large fires and a prominent population of large grazing herbivores (especially water buffalo) are important factors in terms of determining the advance of the *Acacia nilotica* stand into savanna in Baluran National Park. Although a single hot fire is likely to promote *A. nilotica*, it is a frequent fire which may control *A. nilotica* as the species is sensitive to fire when very young. However seed is likely to stay viable in soil for long periods, so managers may need to burn several times (with hot fire) to deplete the soil seed bank. Indeed, several characteristics *of A. nilotica* that facilitates the species to become dominant, namely: its ability to withstand fire and resprout; its ability to persist as potential regeneration following disturbances in a form of soil seed bank; and its possible endozoochory ability as shown in its vast germination in the seeds that remain inside dung/stools. Hence water buffalo (*Bubalus* sp.) may have prominently facilitated dispersal of *A. nilotica* in the savanna. These results further confirmed the important of understanding the seed biology and characters of invasive alien species (IAS) in native community and also confirmed the important roles of seed dispersal through animal digestive systems.

Chapter 7 described dynamics of savanna and forest as important habitat for the endemic and endangered Bali Starling (*Leucopsar rothschildi*) in Bali Barat National Park (BBNP) through the application of remote sensing and field study. Results presented in this chapter suggest that Bali Starling is mostly found in and near to vegetation borders between savanna-forest, savanna-cropland, savanna-shrub land, settlement-cropland, and forest-shrub and. Forest and savanna are important land covers in BBNP and their size dynamics is somewhat related to changes in other types of land cover/uses; nevertheless forest is still the dominant land cover, comprising > 75% of the national park area. Overall, in Bali Barat National Park, changes in vegetation cover and greenness found across the years in which increasing cover of woody plants is the general trend, with savanna areas decreasing in size. However, although the Cekik area had plant species that has been known to be able to provide shelter and food for Bali Starling (as was Brumbun), the bird has not been observed to in the area since the 1990s. This phenomenon may be caused by increased human population and associated infrastructure and land use (dwellings, roads, croplands) in the areas adjacent to BBNP, thereby decreasing available Bali starling habitat, as well as increasing the risk of poaching. This is especially true of the Cekik - Gilimanuk area which has seen a major increase in human-dominated land uses, mainly through conversion of savanna. These results further confirmed the importance of examining habitat patterns of endemic bird within a landscape that are influenced by multiple factors that interact in space and time.

8.3. ORIGINAL CONTRIBUTIONS

The original contributions of this study were the:

- Provision of the first quantitative analysis of plant communities of savanna in East Java, Bali and Lombok Islands and also contribution to the availability of a set of plant species lists of savanna of the wetter parts of the Indonesian Archipelago.
- Quantification of fires in savannas across Indonesia with special emphasis in East Java and Sumba Islands.
- Provision of the first quantitative and spatial analysis of the shift from a *Ziziphus mauritiana* savanna to a homogeneous stand of invasive exotic prickly acacia (*Acacia nilotica*) in a tropical system.
- Clarification of the role of herbivory and fire in facilitating the invasion of prickly acacia (*Acacia nilotica*) in a tropical system.
- Formulation of a conceptual model of the variations in ecological processes that form and maintain savanna in the wetter parts of the Indonesian Archipelago.
- Documentation of habitat dynamics of the endemic Bali Starling (*Leucopsar rotschildi*) in its last remaining original habitat, the Bali Barat National Park. The technique used here can be applied to other savanna which hosts other endemic species to assist in management and conservation of the species and its savanna ecosystem.

8.4. DISCUSSION

8.4.1. Indonesian Tropical Savanna: Similarities with Savannas in Other Regions

Tropical savannas cover over 20% of the Earth's land surface, with the largest coverage in Africa, Australia and South America. Only approximately 10% of savannas on earth occur in India and South-East Asia (Bond and Wilgen 1996; Furley 2004; 1991). Relative to Africa, Australia and South America, little has been written on Asian savannas (Ratnam *et al.* 2016). Floristic descriptions are of interest in comparing similarities and differences within Indonesian savannas and between them and other regions. Based on the MODIS-derived MCD12Q1 mapping product for Asia, countries with proportionally large 'savanna' extents include: Timor-Leste (38.5%, mostly woody savanna); China (38.4%, mostly grasslands); Afghanistan (33.4%, grasslands); Cambodia (30.9% mostly woody savanna); Myanmar (26.9%, mostly woody savanna); and Nepal (24.6%, grasslands) (Beatty *et al.* 2015). Savanna vegetation is also found in Bangladesh, Bhutan, India, Indonesia, Laos, Pakistan, Papua New Guinea, Philippines, South Korea, Taiwan and Thailand (Beatty *et al.* 2015). Tropical savanna ecosystems in South East Asia have been scantly studied (Bird *et al.* 2005; Kurz 1876; Ratnam *et al.* 2016; Stott 1991). Stott (1991), described biogeographically and ecologically what they called savanna forest and open grasslands in mainland South-East Asia however, his study did not include Indonesia Archipelago. Savannas in Indonesia are found across the archipelago over a range of climate and soils (Goltenboth *et al.* 2006; Monk *et al.* 2000; Whitten *et al.* 1996). Surprisingly, in the Indonesian national vegetation classification and field studies, savanna itself is rarely mentioned and described, unlike other type of ecosystems (Backer and van den Brink 1963; Kartawinata 2013; Monk *et al.* 2000; van

Steenis 1972; Whitten *et al.* 1996). Therefore, positioning Indonesia in the world's savanna literature may not be a trivial task as the building blocks are yet to be established but is attemped here based on results from this thesis.

The climatic conditions of Indonesian savanna are generally comparable to those found in other regions such as in Asia, Africa, South America and Australia certainly exist. Savannas in Indonesia occur in regions of seasonal rainfall, but across a wide gradient in annual rainfall (2,940 mm – 800 mm) (Monk *et al.* 2000; Whitten *et al.* 1996). Higher rainfalls occur in the western side of the archipelago and tend to decrease moving to the east. Dry seasons are also longer (more than six months) in the eastern Indonesia. South-eastern parts of Indonesia, such as Sumba Island, has eight months of dry seasons with annual precipitation ranges of 800 – 900 mm. African savannas also occur under markedly seasonal rainfall conditions. Dry seasons last more than six months and tend to increase in length with distance from the equator, and annual precipitation ranges between 600 to 1,200 mm with pronounced inter-annual variation (Beatty *et al.* 2015).

In terms of topography, savannas in Indonesia occur in varied topographical settings across the landscape. Savanna can occupy flat plains such as in Bekol Savanna in Baluran National Park, East Java, but it can also occur in hilly or undulating landscapes of a volcanic mountain slope as observed in the Rinjani National Park in Lombok. African savanna and Australian savanna however, are observed largely occurring on flat landscapes of infertile soils (Beatty *et al.* 2015). In Thailand, open savanna sites are usually topographically drier, such as plateau in the rain shadow of mountain ranges (Kurz 1876). This type of landscape is also occurring in Indonesia, such as Baluran National Park.

In terms of species composition, comparable configurations are also observed. Similarities at the genus and species level were found among the savannas in different regions. Tropical savanna ecosystems in South East Asia, especially India, Burma and Thailand, have been described by Kurz (1876). Common species in these savannas were *Imperata cylindrica*, *Panicum repens*, *Saccharum spontaneum*, *Sorghum halepense*, *Vitiveria zizanoides*, *Eupatorium odoratum, Acacia cathecu, Acacia siamensis, Careya arborea*, and *Pterocarpus macrocarpus*. Several of these species such as *Imperata cylindrica*, *Panicum repens*, *Saccharum spontaneum* and *Eupatorium odoratum* (*Chromolaena odorata*), were also found in the savannas I studied in Indonesia. It is these common savanna species (and genera) which suggest that some savanna in Indonesia may be long standing and persistent (Ratnam *et al.* 2011; Ratnam *et al.* 2016). In addition, at the genus level, Acacia is also found in Indonesian savannas (*Acacia nilotica, Acacia leucophlea* and *Acacia tomentosa*) (Monk *et al.* 2000; Whitten *et al.* 1996). Palm savannas were also observed in Indonesia with species such as the *Corypha utan* in Baluran National Park and *Borassus flabellifer* in Bali Barat National Park and in Kupang East Nusa Tenggara (NTT) (Monk *et al.* 2000; Whitten *et al.* 1996). The same genus *Borassus* (*Borassus aethiopum*) also characterised the humid Lamto savanna of the Cote d'Ivoire in West Africa (Barot *et al.* 1999). In Baluran savanna in East Java as well as in drier parts of Indonesia, such as Komodo, Rinca and Flores Islands, East Nusa Tenggara, *Ziziphus mauritiana* was observed to be present. The same genus (*Ziziphus*) was also found in the Nguru savanna in Nigeria (Hess *et al.* 1996). The Llanos of Colombia and Venezuela is an ecosystem of flammable savanna vegetation because of the presence of highly flammable C3 & C4 grasses (Beatty *et al.* 2015; Huber 1995). In the Llanos, several genus that share similarities with Indonesian savannas are *Andropogon, Paspalum, Mimosa* and *Desmodium*. Meanwhile Australian savannas share a similar problem of the invasive exotic species *Acacia nilotica* (Dhileepan 2009; Radford *et al.* 2001b) with Baluran National Park East Java Indonesia.

8.4.2. Dynamics of a Savanna Landscape: How are Savannas Formed and Maintained?

Savanna and forest are often considered as alternative states within a state and transition framework (Gillson and Ekblom 2009; Twidwell *et al.* 2013). Previous findings and evidence suggested that savanna is not a stable state, instead it is a transition, or also an alternative stable state, where climate, fire, and soil nutrient stocks are inter-related creating dynamic coexistence between the savanna and forest (de L. Dantas *et al.* 2013; Hirota *et al.* 2011; Hoffmann *et al.* 2012a; Silva *et al.* 2013; Silva 2014). Coexistence of trees and grasses is an essential part of the savanna dynamic, which can be a human-created and maintained environment. Grazing and fires have had a profound influence on the vegetation in savannas (Carlsson 2005).

Fires can be of natural cause or anthropogenic in origin, and in the case of NTT human-induced fires are the most prominent (Beatty *et al.* 2015; Tacconi and Ruchiat 2006). Seasonally dry tropical forest (SDTF), climate and fuel loads may be related in terms of significant fires. This could mean that wetter wet seasons will mean more fuel, but it also could be that fuel builds up over time so is independent of climate. Therefore, it could be worth exploring the interactions between climate/weather and fuel in the context of SDTF and tropical savanna ecosystems. Repeating high intensity fire events in SDTF as observed in Baluran National Park resulted in floristic shifting of SDTF towards that of a savanna formation. Fire and grazing could maintain this savanna formation. Without fire, most likely savannas in Indonesia especially in the wetter parts of the Archipelago will most likely convert back to closed canopy and secondary regrowth forest, such as observed in Pangandaran savanna (Rosleine and Suzuki 2013). In this research, this view is also evidenced in type/habitat of woody species found in my savannas (often forest edge or pioneer species).

Grazing helps reduce tree cover when herbivores consume tree seedlings (Carlsson 2005). However, grazing could also promote increase in woody cover and spread of invasive species through endozoochory (Anderson *et al.* 2014) or selective grazing. Intense grazing would cause a drop off in grass biomass which then leads to reduced fuel load that makes fire less severe and, thus, less damaging to trees, therefore resulting in an increase in woody vegetation (van Langevelde *et al.* 2003).

One of the prominent threat to savanna formation is exotic invasive species (Ratnam *et al.* 2016). Figure 1 describe alternative scenario of other stable state options as an impact of invasive alien species and its interaction with herbivores. This is true for IAS *Acacia nilotica* in Baluran. This invasive species may be different with other invasive species that were found in other savannas studied in this research which tended to increase in the absence of burning. Mature *A. nilotica* can withstand fire and sometimes its establishment is assisted by fire; however the young seedling stage of *A. nilotica* is sensitive to fire. This exotic species that was deliberately introduced for fire breaks in Baluran National Park therefore may potentially cause a shift from savanna to a homogenous thicket of woody trees. This is where restoration is needed (depict in figure 1 as dash green arrow).

There is now abundance literature regarding woody thickening and encroachment into savanna and grassland (Belay *et al.* 2013; Cabral *et al.* 2003; Mitchard and Flintrop 2013; Smit *et al.* 2016). Savanna ecosystems are a vital part of the landscape such as in Africa and elsewhere which maintain the livelihoods of millions of people (Devine *et al.* 2017; Mitchard and Flintrop 2013) and although savanna woody encroachment is important phenomenon, its causes are extensively debated (Devine *et al.* 2017). Across Africa, rainfall is the mainly essential determinant of maximum woody cover whereas fire and herbivory interact to decrease woody cover below the maximum (Devine *et al.* 2017). Another view proposed is that woody encroachment is most likely determined by CO_2 enrichment (Smit and Prins 2015; Stevens *et al.* 2017). In Baluran, it is clear that grazing animals such as water buffalo helps to spread the seeds of the invasive alien species (IAS) *Acacia nilotica* and create conditions where woody encroachment is threatening the savanna ecosystem. In Baluran, the juvenile stage of the thorny acacia (*A. nilotica*) cannot be consumed by grazers and later in their mature phase they create thorny thicket formation that limits the chances of grasses species to grow. This woody encroachment in Baluran savannas could further threaten large mammals such as Barking deer (*Muntiacus muntjak*), Sambar deer (*Cervus unicolor*) and the endangered Javan banteng (*Bos javanicus* subsp. *javanicus*) due to the loss of browsing and grazing fields (Sabarno 2002). It is at this stage that rehabilitation to restore the savanna formation is needed.

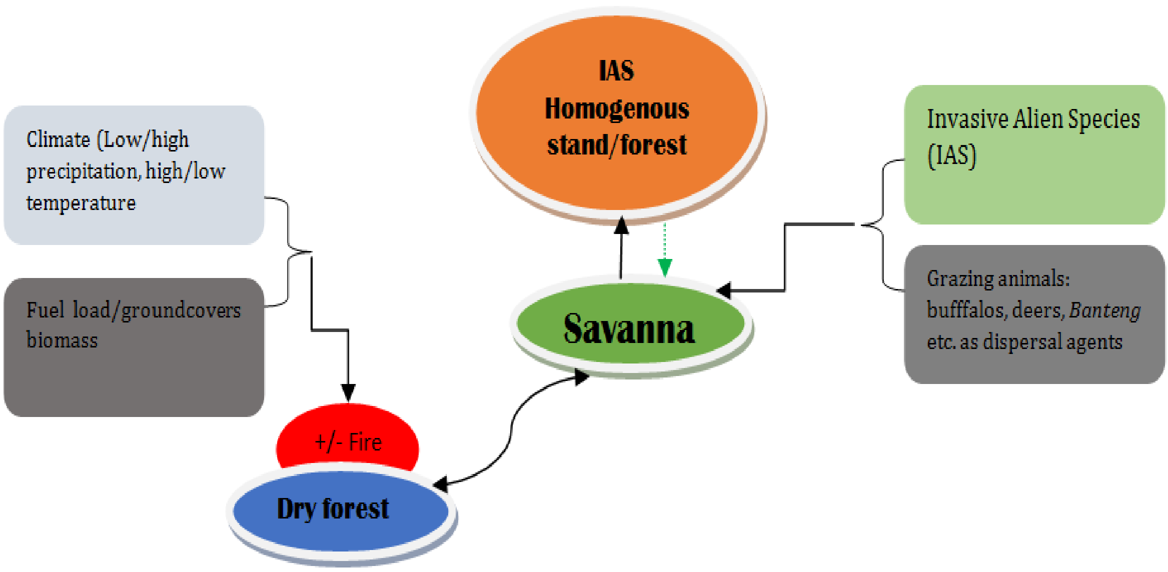


Figure 1. Conceptual model for savanna ecosystem in Indonesia showing potential ecosystem transitions. Green arrow indicates stage where rehabilitation to restore the savanna formation is needed.

Beerling and Osborne (2006) proposed hypotheses for the origin of the world's savanna biome. They concluded that fire accelerates forest loss and C4 grassland expansion through multiple positive feedback loops that promote further drought and more fire. Tree recruitment is also limited by the low CO_2 environment which also amplifies the loop cycles, and allows C4 grasses to thrive and greatly increase ecosystem flammability. Continued intensification of land use could enhance or moderate the network of feedbacks that have initiated, promoted and sustained savannas for millions of years (Beerling and Osborne 2006). However, there are other views regarding the origin of savannas. Currently there are three categories regarding origins of savannas namely climatic savanna, edaphic savanna and derived savanna (Ford 2010; Murphy 2008; Scheiter 2008). From their result of simulating 'fire-off' ecosystem distribution, Bond *et al.* (2005) showed that vast areas of humid C4 grasslands and savannas, especially in South America and Africa, have the climate potential to form forests. The non-cultivated areas in SE Asia are characterized by a mosaic of open forests and dense evergreen rain forests. A characteristic vegetation type below 900 m a.s.l., particularly in Thailand, Laos, Cambodia and Vietnam, is an open deciduous dipterocarp forest, with the ground covered by grasses. It is referred to as 'forêt claire' by French-speaking botanists (Backéus 1992). Dipterocarp 'forêts claires' or 'savanna forest' in northern Thailand are considered to have an edaphic origin, as they are found on poor soil (Backéus 1992; Stott 1990). They have since been much extended by man and are maintained through regular burning (Backéus 1992). In terms of a derived savanna, several studies (Hoffmann *et al.* 2012c; Monk *et al.* 2000; Ratnam *et al.* 2016; van Steenis 1972) have shown the existence of savannas that originally were dry forest, monsoon forest or secondary forest that are frequently disturbed by fires. The grasslands of Java are considered by van Steenis (1972) as anthropogenic and are in various seral stages. This condition is similar for Malaysia (Backéus 1992). Eden (1974) reviewed that the extent formation of savanna and grasslands in Papua, is partly man-made, resulting from combined effects of shifting cultivation and burning. Similar to other savannas and grasslands in Asia, their maintenance is also through regular fires.

Indeed, savanna ecosystems in Southeast Asia have long been regarded as anthropogenic, being derived from tropical forests and maintained via ongoing human manipulation, primarily clearing, grazing and/or burning (Ratnam *et al.* 2011; Solbrig *et al.* 1996; Stott 1990). This view has certainly been broadly reported for Indonesian savannas (Goltenboth *et al.* 2006; Whitten *et al.* 1996). However a recent review by Ratnam *et al*. (2016) questions this assumption and points to antiquity of some Southeast Asian savanna. Evidence for this comes from: 1) fossil history and phylogenetic data showing existence of savanna species of plants and animals in the region before humans; 2) dominance by species with adaptations to withstand repeated fire and/or grazing; and 3) climatic consistencies with savannas of other continents. One of the key findings in this research is that, of the four savannas studied, only savanna in Baluran National Park features typical savanna-type tree species (*Ziziphus mauritiana*). Savanna in Baluran also shows adaptations to herbivory (thorns) and drought (small leaves, open architecture) and suggest it may be a relatively old savanna (*sensu* Ratnam *et al*. 2016). Even the invasive species *Acacia nilotica* and *Azadirachta indica* which present in Baluran are typical savanna species (Dhileepan 2009; Radford *et al.* 2001b; Swaine *et al.* 1992). Other evidence of the relative antiquity of the Baluran savanna are the dominance of C4 grasses, its open structure and persistence of native forbs, as well as the long-term presence of indigenous grazing ungulates (e.g. *Bos javanicus*).

Generally for Asia and Indonesia in particular, reliable vegetation and associated land use mapping that precisely show the distribution of savannas and savanna-like formations are lacking. Bearing in mind the definition of savannas as grasslands with varying densities of tree cover, at present there is no existing detailed classification or dependable fine-scale mapping of the distribution of Asian mixed tree-grass systems (Beatty *et al.* 2015; Ratnam *et al.* 2011). The lack of recognition of savannas in the Asian and Southeast Asian region reflects strong historical "forest" bias in the available vegetation classification system (Sankaran and Ratnam 2013). As a consequence, it has led to the perception that most of the open vegetation and savanna-like formations in this region are categorized as derived savanna, which assumes that these savannas were originally forests, and that anthropogenic disturbances such as burning and grazing have converted them to savannas (Sankaran and Ratnam 2013). Ratnam *et al.* (2016) reviewed that the paleo-historical evidence suggests that the savannas of Asia have existed for at least 1 million years, long before widespread landscape modification by humans. They also suggest that at least there are three distinct Asian savanna communities with distinct functional ecologies consistent with fire- and herbivory-driven community assembly, namely deciduous broadleaf savannas, deciduous fine-leafed and spiny savannas, and evergreen pine savannas.

Ratnam *et al.* (2016) have shown that there is a persisting ideological misperception of Asian savannas as 'degraded forest' where in many of the Asian regions, forests and savannas occur as a mosaic of alternate states within a landscape in a lowland areas managed by humans for thousands of years. I found a similar situation exists in Mount Rinjani National Park. However this savanna occurs in highland areas mixed with mosaics or "small islands" of forest. The officials at the Rinjani National Park office consistently view these savannas as a sign of 'environmental decline' which need to be rehabilitated or restored to a forest state (Budiono, 2014, personal communication). This so called “rehabilitation” has been implemented in the national park and may at least partly explain why most of the woody species present in this savanna are typical forest edge and pioneer species. Indeed in many Asian countries the officially formal perspective is, in fact, a ongoing inheritance of the Dutch colonial authorities’ misunderstanding of the landscape as representing an ‘unwanted’ spread of grasslands and as a sign of environmental decline (Dove 2004). Western societies ‘idealize’ forest cover: hence, national and international agencies view grasslands not as a common class of land cover but as a ‘development problem’ (Caillault *et al.* 2014; Dove 2004). This has reinforced significant research efforts focusing on how to prevent the creation of grasslands and on how to ‘rehabilitate’ grassland (Dove 2004). The Indonesian government and their land management agencies tend towards this view, even though the local people continue to regard grasslands as intrinsic to their livelihoods (Tacconi and Ruchiat 2006). Savannas and grasslands are ecologically important ecosystems and in some cases it is their conservation rather than ‘rehabilitation’ that needs to be considered (Tacconi and Ruchiat 2006). Indeed, the threat to the persistence of savannas in Asia include afforestation policies, mismanagement of fire and herbivory, alien woody encroachment, and future climate uncertainty (Ratnam *et al.* 2016).

8.4.3. People in Savanna Landscapes: Fire as a Tool for Sustainability

Savannas landscape sustain about 10% of the human population despite only, occupying one-sixth of the land surface, and while rates of land use alteration are uncertain, savannas may be experiencing double the rate of conversion as that for tropical forests (Beatty *et al.* 2015; White *et al.* 2000). Savannas and humans have a long history of interdependency where pre-humans living in East Africa 4.4 million years ago inhabited savannas or grassy plains with sparse trees and shrubs (SNS 2010). For thousands of years, Indigenous peoples around the globe have used fire as a land management tool (Bloesch 1999; Butler *et al.* 2014). Savanna is the most fire-prone ecosystem, and under current patterns of human livelihood and land use, most fires are started by people (Beatty *et al.* 2015; Bond and Keeley 2005). Savanna fire emissions are mainly sourced from Africa, contributing roughly 71% of all savanna CO_2 emissions, followed by South America (12%), Australia (7.3%), South East and Equatorial Asia (5.9%) (Beatty *et al.* 2015; UNU 2015). As human populations increase and become increasingly urbanised, much of the indigenous fire management knowledge has broken down or been interrupted, causing patterns of burning (and indeed fire regimes) to change (Beatty *et al.* 2015; Tacconi and Vayda 2006). Anthropogenic fires have been vital in maintaining many African savannas over the last 1.5 million years with humans possessing considerable control over fire regimes and biomass burning for at least 400,000 years (Beatty *et al.* 2015; Bond and Zaloumis 2016; Shaffer 2010). Regional patterns of burning in African savannas are significantly affected by the higher rural population densities and different land use practices (Bistinas *et al.* 2013). Fire in African savannas has long been depicted as a degrading force that has been aligned to deforestation, savannisation and desertification (Caillault *et al.* 2014). In South America, savannas are experiencing major threats as the current growth of agriculture has caused the swift conversion of 71% of savannas to croplands and 5% to urban areas (Beatty *et al.* 2015; White *et al.* 2000).

Fire contributes to livelihoods in the savanna-based and swidden agricultural systems; this slash and burn technique is applied in both tropical forest landscape (in a form of swidden cultivation/shifting cultivation), and also in more open, less woody (derived or natural) savanna landscape in the Asian region. Swidden cultivation has traditionally and continues to be widely used in South East Asia and details of these practices vary according to cultural settings (Beatty *et al.* 2015). Swidden cultivation in Southeast Asia is described by Mertz *et al.* (2009) as system of land management that involves a natural fallowing phase which takes longer than the cultivation phase of annual crops, this fallow stage is long enough for woody vegetation to dominates the site and then cleared using fire. These cultivated plots usually then are used for approximately five years until they become unproductive; these processes are then repeated at another site. In a forested system, which includes less productive semi-arid areas, fallow periods of 10 to 25 years are considered to be sustainable (Beatty *et al.* 2015; Fui *et al.* 2012; Mertz *et al.* 2009). In several aspects, swidden or shifting cultivation is related to soil fertility (Siregar 2006). Population growth and pressures on the ecosystems from agriculture and industries in Asia has been affecting swidden cultivation (Fui *et al.* 2012). Fallow periods have become as short as 3 to 5 years in various situations, with ensuing impacts on declining soil fertility (Beatty *et al.* 2015; Fui *et al.* 2012; Setyawan 2010). The savannas studied in this thesis were all in national parks; however some level or type of swidden agriculture or 'agricultural burning' may have taken place in the past.

Following the large fire events in 1997-98, the Indonesian government was under strong international pressure, especially from ASEAN countries, to limit the recurrence of uncontrolled fires (Aiken 2004; Tacconi and Ruchiat 2006). This comes under the regional Agreement on Transboundary Haze Pollution, which then gave birth to the Indonesian National Regulation 4/2001 which enforces a zero burning policy, including in grasslands/savannas. This policy also focusses on fire suppression and crisis management instead of fire management (Potter 1996; Tacconi and Ruchiat

2006). Varma (2003) argue that slash and burn methods that are used in traditional shifting cultivation is thought to be the underlying cause of the catastrophic fire events. However, this view is not entirely accepted (Beatty *et al.* 2015). Tacconi and Vayda (2006) challenge the view with several underlying arguments. Firstly, Varma (2003) defines slash and burn agriculture too broadly and misrepresents the nature and causes of the 1997–98 fires by virtue of attributing them entirely to slash and burn agriculture, whereas, in fact, roading and logging by timber companies over large areas may be more important. Secondly, the economic assessment of the costs of the fires was also problematic. The zero burning policy has failed to accommodate the specific characters of the diverse ecosystems of Indonesia. It has also failed in delivering improved environmental and social outcomes (Beatty *et al.* 2015). Evidences of the failure of the zero burning policy have been heavily drawn from studies in eastern Indonesia where the drier climate is coupled with vast landscape of fire-prone savannas and grassland (Djoeroemana and Myers 2000; Russel-Smith *et al.* 2007; Tacconi and Ruchiat 2006).

Land-use practices in more open, less woody (derived or natural) savanna landscape in the Asian region involves pastoral activities and related burning practices which appear to assume greater importance in the mix of livelihood options (Beatty *et al.* 2015). In eastern Indonesia, particularly islands such as Sumba and Flores, fire is utilized in various livelihood activities such as in agriculture (to clear land, slash and burn), livestock management (to regenerate grassland), and hunting (to locate prey) (Monk *et al.* 2000; Tacconi and Ruchiat 2006). Burning for small landholders in eastern Indonesia, especially in Sumba and Flores, is the most efficient method of land clearing and regenerating grasslands and maintaining grazing land (Tacconi and Ruchiat 2006). In chapter three of this thesis it was found that Sumba Island was the most prominent (compared with other areas of NTT) in terms of fires hotspots and burned area detected from 2000 to 2013. Hotspots were detected in almost all the northern coastal and eastern regions of Sumba Island. Fire hotspots were detected more in savanna country/open vegetation, compared to forests on the island. August (peak of dry season) is the month where the total number of fire hotspots detected and burned areas were at the highest level. Indeed, fire is widely used in swidden agricultural activities and in grassland management in both Flores and Sumba, not unlike elsewhere in the tropics (Monk *et al.* 2000; Tacconi and Ruchiat 2006). Important local knowledge regarding utilization of fire exists in Flores and Sumba. Fire management on these Islands includes establishing fire breaks, and fires are set at dusk when temperatures are cooler allowing for better control and generally lower intensity and, therefore, impact (Kaho 2015; Tacconi and Ruchiat 2006).

There is increasing acknowledgment of the similarity of savannas between northern Australia and East Nusa Tenggara (NTT) Indonesia in terms of climate and fire regime (Beatty *et al.* 2015; Djoeroemana and Myers 2000; Radford *et al.* 2001a) and this has led to the development of joint fire management programs and cooperative approaches, linking practitioners from northern Australia with relevant parties in eastern Indonesia (Russell-Smith *et al.* 2000b). This program aims to adopt methods and experience gained in controlling wildfires in savannas in Australia and Africa to eastern Indonesia. Recent experience in remote north Australia demonstrates reintroduction of traditional, early dry season fire management practices can reduce the amount of biomass burnt by savanna fires, increase and improve habitat for endangered species and reduce emissions, including CO_2 (Russel-Smith *et al.* 2000). However, proposed changes in burning practices to reduce the negative impacts of fire as suggested in the program should account for livelihood strategies as well as potential negative environmental impacts (Tacconi and Ruchiat 2006). Changing of burning time to the early dry season may not be suitable for NTT (Monk *et al.* 2000) as it may negatively affect grasslands if a second burn is needed to stimulate regrowth for cattle grazing. Burning before the dry season is not recommended because it can reduce the amount of grass available during the peak of the dry season (Tacconi and Ruchiat 2006). Fire contributes to livelihoods and the sustainability of savanna-based and swidden agricultural systems in Indonesia, however, to become effective, future government regulations should formulated in line with the geographic variation in fuels, climate and land use, alongside the human livelihood dimension, across Indonesia (Tacconi and Ruchiat 2006). Improvement in fire management would be expected to enhance biodiversity protection in savanna, decrease emissions and support efforts to avoid savanna woodland degradation, improve agricultural production, weed management and pasture and livestock quality, improve food security, improve health indicators, reduce human conflict, promote infrastructure security, and assist in climate change adaptation as well as disaster resilience (Beatty *et al.* 2015; UNU 2015).

8.4.4. Endemic Wildlife Species in Savanna Landscape

Savannas support various species of wildlife. African savannas are characterized by many charismatic and endemic large mammal groups (Beatty *et al.* 2015; Dublin 1995; Fjeldsa *et al.* 2004). Llanos of Venezuala and Colombia are rich in biodiversity. There are no endemic birds limited to the Llanos eco-region, and only two mammals: the marsupial *Monodelphis orinoci* and the edentate *Dasypus sabanicola* (Beatty *et al.* 2015; Eisenberg and Polisar 1999). Fossil evidence suggest that savannas present in Asia before human arrival and that endemic species with life histories adapted to savanna environments, exist (Ratnam *et al.* 2016). Tropical savanna ecosystems in South East Asia, especially India, Burma and Thailand, have been described by Kurz

(1876). Where such savanna occur, they provide important habitat for large browsing and grazing mammals such as Barking deer (*Muntiacus muntjak*), Sambar deer (*Cervus unicolor*) and Banteng (*Bos javanicus*) (Kurz 1876).

A number of ruminant mammals are endemic to savannas in Asia. Several Bovinae (Blue bull, *Boselaphus tragocamelus*) and Cervidae (Rusa, Rusa *timorensis*) are associated with fine-leafed and spiny savannahs in South and Southeast Asia. Whereas, other Bovidae (Banteng, *Bos javanica;* four horned antelope, *Tetracerus quadricornus*) in South and Southeast Asia are associated with deciduous broadleaf savannas (Ratnam *et al.* 2016). Two iconic species of South and Southeast Asia, the Indian and Green peafowls (*Pavo* spp.), prefer deciduous broadleaf savannas to forests (Hernowo 1995; Ratnam *et al.* 2016). Tropical savanna and dry forest in Indonesia are an important type of ecosystems that supports various endemic wildlife of Indonesia and some of them are under serious threat and have high conservation status according to IUCN categories, such as Javan banteng (*Bos javanicus* subsp*. javanicus*) in the savanna of Baluran National Park, the endemic Bali Starling (*Leucopsar rotschildi*) in Bali Barat National Park's savanna, and the Komodo dragon (*Varanus komodoensis*) which is endemic to the Komodo Islands and also west Flores of East Nusa Tenggara (NTT).

Baluran National Park is home to 26 species of mammal, including the endangered Banteng, Dhole, Indian muntjac, Java mouse-deer, Fishing cat, and Leopard and Javan lutung. Bird species in the park include the Green Peafowl, Red Jungle fowl, Malabar Pied Hornbill, Rhinoceros Hornbill and Lesser Adjutant (Baluran 2010; Hernowo 1995; Hernowo *et al.* 2011). Until 2010 there had been 155 species of bird recorded in the park; this was number was revised in in 2012 to 196 birds (BTNB 2010). The savanna of Baluran National Park has significant role in supporting a diversity of plants and big mammals, including herbivores such as the Javan banteng

(*Bos javanicus* subsp*. javanicus*), a species of wild cattle found in Java and Bali which is categorized as an endangered species in the IUCN Red List (IUCN 2008; Sabarno 2002; Suhadi 2009; Widayanti 2010). The banteng population in Baluran decreased from 338 in 1996 to just 26 in 2012 (Baluran 2010). A local breeding program within Baluran National Park (Figure 2) has been successful in captive breeding and releasing Banteng into the wild, where in 2014, it was estimated that the number of banteng increased to 35 in the wild within the national park (Sabarno, 2014, personal communication).

Figure 2. Male 'Javan banteng' (*Bos javanicus* subsp*. javanicus*) at the captive breeding program inside Baluran National Park, October 2014.

Just across the Bali Strait from Baluran, another endangered species occurs in Bali Barat. Bali Barat National Park is the last sanctuary for the Bali Starling (Figure 3). The critically endangered Bali starling (*Leucopsar rothschildi*) is the only endemic bird found in Bali. Stresemann (1912) collected and described the first Bali starling known to science near Bubunan, Bali. Bali Starling original habitat in Bali was described as 'dry savanna and shrub woodlands', but also 'tall and dense forest' in the 1920's (van der

Paardt 1926). Results from chapter seven of this thesis showed that Bali Starling range in Bali Island shrunk to the fire-induced open shrub and savanna woodland, found below an elevation of 150–175 m in the north-east part of peninsular Prapat Agung of Bali Barat National Park, confirming the result of van Balen *et al.* (2000). Bali Starling prefers savanna and open vegetation but also may need nearby forest for breeding. One of the important findings of this research was also that there was a pattern of increasing tree density and woody cover observed in Bali Barat National Park. This phenomenon may have important effect on the Bali Starling. Decreases in savanna/grassland could reduce options of food resources for Bali Starling. Bali starlings are omnivorous but feed their young an insectivorous diet (Seibels *et al.* 1997). Birds of the Sturnidae family are usually symbiotic with cattle which feed on the insects living on the surface body of the cattle; therefore such birds usually prefer more open grazing lands, savanna, or open plantations that are adjacent to a nearby forest (MacKinnon *et al.* 1998). Where there is such remaining savanna/grassland in the national park, the Bali Starling still may have to compete with other birds. A recent survey by Widodo (2014) did not find any Bali Starling in these areas (although they had found them there in 2010). It was noted that Bali Starling compete strongly and are often defeated by Raja Udang, also known as collared kingfisher or *Todiramphus chloris* (Widodo 2014).

Land use change and poaching are significant threats for Bali Starling existence (Dirgayusa *et al.* 2000). *In situ* conservation have been conducted since 1994 in Bali Barat National Park through Bali Starling Project (BSP) which has included a successful captive breeding program to supplement the wild populations of the Bali Starlings (Earnhardt *et al.* 2009; van Balen *et al.* 2000). The dangerous effects of a small population size, the complexity of the law enforcement system and harsh local climatic conditions have so far barred the Bali starling from emerging from the verge of extinction (van Balen *et al.* 2000). Despite major achievements in the captive release methods it has not yet resulted in a persistent recovery of the wild population of Bali starlings (Collins and Smith 1994; Collins *et al.* 1998; van Balen and Gepak 1994; van Balen *et al.* 2000). In 1997, the Bali Starling species Recovery Plan was initiated, involving all implementing parties in Indonesia. The plan provides guidelines for continuation of the conservation programme. It comprehensively addresses many aspects of Bali Starling conservation, such as protection in the wild, extension of breeding programmes, habitat restoration and law enforcement (van Balen *et al.* 2000). Prescibed burning to maintain open savanna habitat, as well as sharp forest-savanna vegetation boundaries, may need to given more attention in subsequent recovery plans.

Figure 3. Bali Starling (*Leucopsar rothschildi*) in Bali Barat National Park. Photo courtesy of Mr. Wiryawan, Bali Barat National Park, 2014.

8.4.5. Invasive Alien Plants in Savanna Landscape

Savanna-adapted species are found across the three savanna regions of Asia (Beatty *et al.* 2015; Ratnam *et al.* 2016). Some species in the fine-leafed and spiny savannas may be limited to Asia such as *Acacia leucophloea*, but many characteristic species expand beyond Asia (e.g. *Dichrostachys cinerea, Ziziphus mauritiana, Acacia nilotica*) (Kriticos *et al.* 2003; Radford *et al.* 2001a; Ratnam *et al.* 2016). Exotic invasive species pose serious threats to the persistence of savanna (Ratnam *et al.* 2016). These invasive alien plants include flammable non-native grasses such as African grasses that behave as very aggressive invaders (in southeast Asia and globally) and increase fire intensity and perhaps frequency (Baruch 1996). These African grasses include: *Melinis minutiflora*, which is very successful in invading savannas above 600 metres above sea level; *Hyparrenia rufa*, in lowland savannas with poor soils and a marked dry season; *Panicum maximum*, in humid and relatively fertile areas, and *Brachiaria mutica* in periodically flooded savannas (Baruch 1996). All these species generally occur in the wetter (but not inundated) and/or more fertile habitats of the savanna (Baruch 1996; Beatty *et al.* 2015). In Asia, several exotic species have also been reported as invasive such as *Chromolaena odorata* (Bangladesh, Nepal), *Lantana camara* (Bangladesh, Maldives, Nepal, Pakistan, Philipine), *Mimosa invisa, Mimosa pigra* (Laos, Malaysia), *Mikania micrantha* (Malaysia, Nepal, Philipine), *Leucaena leucocephala* (Maldives, Philipines), *Acacia mangium* (Philipine) and *Eucalyptus camaldulensis* (Pakistan, Philipines) (Pallewatta *et al.* 2003). Most of the species are also found to be invasive in Indonesia as well. In my research, I found that some of these species such as *Lantana camara*, and *Chromolaena odorata* are significant threats to the savanna ecosystem in Indonesia in addition to the more well known woody alien species *Acacia nilotica*. Many invasive woody plants and climbing plants were present in the Bali Barat savanna, which is likely to reflect its proximity to human settlements and activity. Similarly at Alas Purwo, domination by many invasive alien species, especially *Chromolaena odorata*, was recorded (Appendix 1). Both Bali Barat and Alas Purwo savannas are in danger of being transformed into a forest structure due to lack of fires and prevalence of such invasive species.

Savanna in Baluran National Park was introduced with a exotic woody plant species from Africa, *Acacia nilotica,* in the late 1960s, where its original purpose was to create fire breaks to prevent fire spreading from the Baluran savanna to adjacent teak forest (Sabarno 2002). However today *A. nilotica* has spread rapidly and is threatening the existence of Baluran savanna as it has been observed to cause changes in ecosystem from open savannas to a closed canopy of *A. nilotica* in some areas (Djufri 2004; Sutomo *et al.* 2016). In Australian savanna ecosystems study by Radford *et al.* (2001), *A. nilotica* was acknowledged to have major negative impacts on savannas. Its adult trees are fire tolerant and can form thorny thicket formations (Burrows *et al.* 1986). This shift from open savanna to woodland and thicket will likely result in serious environmental impacts, such as changes in herbaceous plant communities, faunal habitat and soil hydrology (Radford *et al.* 2001a).

Eradications efforts to eliminate *A. nilotica* in the savanna ecosystem of Baluran National Park commenced in 1985 with various collaborations with government bodies, as well as the private sector (Djufri 2004; Sabarno 2002). Three methods have been applied namely chemical, mechanical and manual control. However up to 2009 there were no satisfactory results. The areas that were invaded by *A. nilotica* have never been returned back to their original state (savanna) before the invasion using current control efforts (Wahono 2009). From 2011, eradication efforts have been led by the Indonesian Centre for Research and Development of the Forestry Ministry. Basically the method is to apply strong herbicide (triclopyr) solution mix with diesel and then apply it to the bark of *A. nilotica* (stem/stump brushing). After one year, leaves have fallen, bark decayed and broken and subsequently the tree will fall. Soon after, the trees are burnt and grasses planted (Figure 4).

Figure 4. *Acacia nilotica* tree after application of tryclopyr on its stem. Note the existence of *A. nilotica* seedling near the dead tree (yellow square).

However, although this method seems quite promising, it still needs to be assessed for its effectiveness and efficacy. In terms of potential natural regeneration, managers need to be assured that there are no seed stored in the soils around the restoration site. One issue which need to be addressed is persistent soil seed bank (SSB) which I found to be clearly present in the restoration site where it had the highest level of seed viability (as shown in chapter six). The burning prescription used also may accelerate the germination of these soil seed bank in the restoration site, as I found that mild fire may promote germination, especially where seeds are insulated by soil or stools (as per results of my chapter six). Therefore, further soil seed bank assessment also needs to be considered by the Baluran National Park management office as one of the routine variables to measure the success of the restoration works. Soil seed bank assessments may also be important for general monitoring of the savanna ecosystem as it informs managers about potential regeneration and vegetation dynamics. In the future, the eradication program needs to be based on the concept of adaptive management, as it would involve learning by doing and monitoring effects of any intervention to improve management over time.

8.4.6. Approaches in Remote Sensing and Modelling in Recognizing Savanna Ecosystem in Asia

The important role of remote sensing (RS) and geographical information systems (GIS) in ecology, land management and conservation has been recognized (Arno *et al.* 1977; Chuvieco and Congalton 1989; Keane *et al.* 2001; van Wilgen *et al.* 2000; Verlinden and Laamanen 2006). Globally, studies on the use of RS/GIS in savanna ecosystem are also gaining attraction (Banfai and Bowman 2005; Chacón-Moreno 2004; Hudak and Brockett 2004; Sano *et al.* 2010; Stroppiana *et al.* 2003). However, there is still lack of specific RS/GIS studies on Asian savanna ecosystems. Beatty *et al.* (2015) recognized that there is a lack of reliable vegetation and associated land use mapping that accurately depicts the distribution of savannas and savanna-like formations in Asia; they also identified the need for fire mapping and associated emissions data in Asian savannas. In addition, there is also lack of study in modelling Asian savanna dynamics and ecosystem function.

Remote sensing and modelling can be applied to assist in classifying the savanna landscape (Bond *et al.* 2005; Lehmann *et al.* 2011; Stuart *et al.* 2006). Stuart *et al.* (2006) combined data from GPS ground survey with classifications of medium spatial resolution LANDSAT imagery to distinguish variations within neotropical savannas in Belize and to characterize the boundaries between savanna areas and associated forests and wetland communities. At the global scale, the Dynamic Global Vegetation Model (DVGM) and RS/GIS techniques were used by Bond *et al.* (2005) to simulate global distribution of ecosystems in a world where fire is absent. Their results show that vast areas of humid C4 grasslands and savannas, especially in South America, Africa and Asia, have the climate potential to form forests. Eden (1974) conducted overlay of air photographs and combined results from field studies to describe the savanna and grassland vegetation in Papua and their origin. Stibig *et al.* (2003) conducted mapping of tropical forest cover of insular Southeast Asia from coarse resolution satellite images. However in their classification savanna was not mentioned instead they used the term 'sparseland' to named burnt/dry vegetation. Much of what Stibig *et al.* (2003) mapped as sparseland would be considered savanna in eastern Indonesia. This 'sparseland' cover including the vast areas on Sumba Island, NTT. Results from chapter three of this thesis showed that, on Sumba Island, savannas and grasslands are the major landcover. Several vegetation mapping for SEA were also conducted by other researchers (Dong *et al.* 2014; Fuller and Murphy 2006) however they failed to differentiate the savannas. Again as mentioned previously in section 8.4.2., this somewhat showed the phenomenon of 'forest centric' legacy in understanding Asian vegetation (Ratnam *et al.* 2016).

More generalized vegetation or 'land cover' mapping products are available, mostly at global scale, which may be used to broadly describe the regional distribution of 'savanna' types in Asia such as the University of Maryland's MODIS-derived (500 m pixel) land cover product (MCD12Q1; http://glcf.umd.edu/data/lc/) (Beatty *et al.* 2015). The map identify savannas in Timor, Flores and Sumba Islands East Nusa

Tenggara (NTT) Indonesia and Timor Leste but did not recognized savanna in Baluran East Java Indonesia as well as other savannas and grasslands in other regions of Indonesia such as Alas Purwo East Java, Bali Barat, Rinjani Lombok and Tambora Sumbawa. Currently there is insufficient data from Asian savannas to effectively define their climatic domains based on field observation (Ratnam *et al.* 2016). This is where further development of RS/GIS especially in modelling can be of considerable value. In their effort to identify and estimate the most probable geographical extent of Asian savanna distribution, Ratnam *et al.* (2016) mapped the potential extent of Asian savannas based on modelling climate envelopes occupied by African, South American and Australian savannas. The models successfully recreated the observed distribution of savannas in Africa with a high degree of accuracy (92.2% of savanna pixels correctly classified) and to a lesser extent, those of South America (72.5% correctly classified) and Australia (68.2% correctly classified). Then the models developed for each of these continents were used to predict the potential distribution of savannas in Asia. Their results suggest that Asia supports larger areas with climates that are analogous to those occupied by African savannas than either Australian or South American savannas (Ratnam *et al.* 2016). When closely examined, their map of the predicted Asian savannas based on climate envelope of African savannas mostly agrees with results from this thesis and successfully identified vast areas of savannas and grasslands in East Nusa Tenggara (NTT) Indonesia including Sumba, Timor and Flores Islands Indonesia. Moreover, it has also identified the existence of savanna in Baluran National Park areas, as well as several other areas in north-east Java, a place in Indonesia in which is less well known to the scientific community to exhibit savanna ecosystems. Therefore there is support for a climatically-driven distribution of savanna at the broad scale.

Fire monitoring continues to be an area of scientific and management interest within global change research on ecosystem dynamics, particularly through its connection to land cover change assessments (Morisette *et al.* 2005). The assessment of fire regimes across broad territories is carried out efficiently by means of data obtained by remote sensing (Chuvieco and Salas 1996; Devineau *et al.* 2010). Fire monitoring by remote sensing in savanna ecosystems, especially in regions such as Africa and Australia, has been common. Fire monitoring in savanna ecosystems was conducted by Chongo (2007) in Kruger National Park Africa using moderate-resolution imaging Spectroradiometer (MODIS) and calculation of normalised difference infrared index (NDII) and normalised difference vegetation index (NDVI). In Southern Africa, fire scars mapping in savanna landscape using Landsat imagery was done by Hudak and Brockett (2004). In West Africa, Devineau *et al.* (2010) conducted savanna fire regimes assessment using both MODIS active fire data (MOD14A2a and MYD14A2) and burnt area product (MCD45A1) in western Burkina Faso. Application of RS/GIS has also been common in Australia to study fires in its savanna ecosystem (Russell-Smith and Edwards 2006; Stroppiana *et al.* 2003; Wisell 2001). In Asia, active fire detections have been available (although at fairly coarse scale) from MODIS since 2000 (Hyer *et al.* 2013; Justice *et al.* 2002), and some previous applications used the NOAA Advanced Very High Resolution Radiometer (AVHRR) for fire detections (Hyer *et al.* 2013). Other sources of remote sensing data were also used. Fuller and Murphy (2006) used multi-year fire data set derived from the Along Track Scanning Radiometer (ATSR), to analyse temporal and spatial patterns of forest fire for a window covering most of insular Southeast Asia cantered over the island of Borneo. Spatial and temporal distributions of fires detected in the MTSAT WF_ABBA (geostationary satellites operated by the Japanese space agency JAXA) and MODIS products are described by Hyer (2013) for a study region consisting of Indonesia, Malaysia, and nearby environs. However, regional-scale fire mapping data are likely to substantially under-report actual fire extents and frequencies in typical savanna habitats in the Asian region (Beatty *et al.* 2015). Therefore, using high resolution remote sensing is the future, as it can delineate vegetation types potentially at very fine scale. One of the few studies of fires in Asian savannas was done by Fisher *et al.* (2006). Fisher *et al.* (2006) conducted an assessment of biomass burning in grassland and more wooded savanna areas on the East Nusa Tenggara (NTT) islands of Sumba and Flores, but only for two years (2003 and 2004). Landsat images (30 m pixels) were used and it was observed that in two years, 29% of 7,000 km^2 Sumba and 11% of 3,000 km^2 Flores study areas were burnt, with ~90% of all mapped fires being <5 ha extent. Results from chapter three regarding spatial and temporal patterns of fires in tropical savanna in Indonesia (2000 – 2013) also showed more fires were detected in the drier parts of the Indonesian Archipelago, with most burning occurring in East Nusa Tenggara (NTT) Province, particularly Sumba and Timor Islands. This result supplements the work of Fisher *et al.* (2006) and fills the gap in the application of remote sensing for fires detection in Asian savanna landscapes. As argued above, using high resolution remote sensing is the way forward. We need to map fires at finer scale as many fires in savanna, as shown in chapter three, are small, low intensity and patchy, and therefore may not be picked up by MODIS (although MODIS has the advantage of frequency and length of operation).

As concluded by Ratnam *et al.* (2016), the main threats to the persistence of savannas (including in Asia), are mismanagement of fire and herbivory, alien woody encroachment (invasive alien species), afforestation policies and future climate uncertainty associated with the changing climatic regimes. Modelling approaches have been applied to answer some of these problems in savanna-grassland landscapes elsewhere, but is currently lacking in Asia. Savannas occur across most of northern Australia and are extensively used as rangelands (grazing by domestic livestock) (Beatty *et al.* 2015). Modelling the resilience of Australian savanna systems to grazing impacts (herbivores management) was done by Ludwig *et al.* (2001). A process model, SAVANNA, was parameterised to simulate the structure and function of Australian savannas. Simulations were run for 50 years at two levels of grazing to evaluate resistance and then for another 50 years with no grazing to evaluate recovery. These runs predicted that savanna grasslands were more resistant to grazing (changed less) than red-loam woodlands, which recovered relatively slowly from grazing impacts. In terms of climate change regimes, predicted climate change effects on the distribution of meadows in the arid and semi-arid Argentinian Patagonia by 2050 were assessed for change trends and areas of desertification vulnerability using SDM and climate change models by Crego *et al.* (2014). Their result showed that suitable areas for meadows in Argentinian Patagonia will decrease by 7.85% by 2050. In another region, spatial simulation model (LANDIS-II) under conditions of climate change, current fire and alternative management regimes was used by Mairota *et al.* (2014) to develop a scenario maps to examined woody plant encroachment in Mediterranean grassland. Their models point out that around one-third of the grassland area would be impacted by loss, fragmentation and degradation in the next 150 years. In terms of invasive alien species or woody encroachment by alien species in Australian savanna, Kritikos *et al.* (2003) conducted species distribution modelling using CLIMEX and predicted the potential distribution under future climate change scenario for invasive alien species *Acacia nilotica* subspecies *indica* in Australia. Based on their species distribution modelling and climate change projection it is expected that there may be increases in water-use efficiency of the species due to increased atmospheric CO_2 concentrations, allowing it to invade more xeric sites further inland, and increased temperatures. Result of species distribution modelling in chapter five of this thesis show that global climate change is likely to increase the potential distribution of *Acacia nilotica* in Indonesia and considerably increases the area at high risk of invasion by year 2045. This study is one of the first to model and highlight the savanna regions with at highest risk of invasion by a prevalent invasive alien species in tropical environments of Southeast Asia.

8.5. CONCLUSION AND RECOMENDATIONS

Savannas globally are intriguing ecosystems that sustain distinctive communities of pastoral and agro-pastoral people together with wild and domestic herbivores. In summary, the description of the savanna dynamics presented here provide further evidence of the complexity of the savanna ecosystem and its susceptibility to change as a result of changing fire regimes (particularly decreased fire frequency or exclusion) and invasion by invasive species. Results from this study have shown that savanna plant communities in Indonesia are formed and maintained by the interaction between climatic factors, fire regime and grazing, and that dispersal of invasive alien plant seeds through herbivory, combined with burning, could change the savanna ecosystem to an alternative state. This ecosystem shifting not only has significant effects on the overall plant composition, but also affects the savanna both ecologically and socio-economically. It could also carry consequence for endemic wildlife inhabiting the savanna.

Time frame and logistical issues, however, limit the conclusions to be drawn from this research to some degree. The vegetation surveys were only carried out in late dry season rather than early in the wet season and may not be representative of conditions throughout the year (likely meaning lower diversity and grass cover are reported). Locations of the study sites were also separated by seas and located on different islands; therefore the budget for mobility and time constraints became very crucial. I did not have a budget to study more savannas across a greater area of Indonesia. Data collection over a longer time period and perhaps establishing permanent sample plots and regular observation may provide a more complete understanding of the patterns of savanna vegetation dynamics and the invasion by exotic species (IAS). The germination experiment in the laboratory to simulate effect of fire and IAS seed dispersal was also performed for a limited time period and with few variables hence may not represent stochastic nature of the ecosystem.

A greater understanding of the possible ecosystem processes driving the dynamics of the savannas will assist in the formulation of successful savanna management strategy. Options for further work to build on the findings reported here could be obtained by: 1) continuing and extending the spatial and temporal vegetation surveys to better define the vegetation dynamics between the alternate states; and 2) long-term monitoring, through permanent sample plots, together with controlled experiments and remote sensing-modelling approaches of the ecosystem function would allow further exploration of the key savanna ecosystem processes and patterns. In terms of remote sensing, monitoring of land use change and fire regimes changes are needed and this would benefit by availability of higher resolution spatial imagery products (in the Asian context). In terms of modelling, modellers are increasingly using ensemble (also known as consensus) modelling techniques that improve prediction accuracy over single models (Crego *et al.* 2014). Therefore application of multiple modelling approaches to identify options for savanna ecosystem restoration would be beneficial. Secondly, extending the combined modelling-remote sensing approach to more actively study conservation of endemic species of plants and wildlife species that characterize the savannas. Addressing data shortage in habitat patterns within endemic species distribution is important for conservation managers developing conservation management strategies. In ecosystems where threatening processes are still acting on a population, such research should be a high priority for conservation managers.

APPENDIX 1: List of all species found in chapter 2

List of all species found in plots for chapter 2, showing their taxonomic, life-form and habitat characteristics. The location and number of plots in which the species was found is also indicated. AP = Alas Purwo: GR = Rinjani; BB = Bali Barat; BA = Baluran.

Species	Family (Tribes for grasses)	Photosynthetic pathway (*based on genus)	Life-form	Native in study region#	Habitat	AP Sites	GR Sites	BB Sites	BA Sites
Acacia nilotica (syn *Vachellia nilotica)*	Fabaceae		Tree, thorny	No (Africa-Arabia)	Savanna, open forests	0	0	0	4
Ageratum conyzoides	Asteraceae		Forb	No (Americas)	Forest, disturbed	0	0	0	1
Albizia chinensis	Fabaceae		Tree	Yes	Dry forest, forest margins and savanna	0	0	2	0
Antidesma bunius	Phyllanthaceae		Tree	Yes	Forests	0	0	4	0
Arundinella setosa	Poaceae (Arundinelleae)	C4*	Grass (perennial)	Yes	Open forest, forest margins and savanna	10	0	0	0
Azadirachta indica	Meliaceae		Tree	Likely no (India)	Savanna, cultivated (drought tolerant)	0	0	0	1
Borassus flabellifer	Arecaceae		Tree (palm)	Yes	Savanna, open forests	0	0	7	0
Bridelia stipularis	Phyllanthaceae		Shrub	Yes	Mostly forests	0	0	5	0
Calliandra calothyrsus	Fabaceae		Small tree	No (Americas)	Forests, cultivated	0	0	2	0
Callicarpa pedunculata	Lamiaceae		Shrub	Yes	Forests, disturbed	0	0	1	0
Calotropis gigantea	Apocynaceae		Shrub	Yes	Dry forests, cultivated	0	0	1	0
Chromolaena odorata	Asteraceae		Shrub	No (Nth America)	Widespread weed, forests, edges	9	0	5	0
Cordia bantamensis	Boraginaceae		Small tree	Likely yes	Dry forests	0	0	0	1
Cyperus brevifolius	Cyperaceae	C4*	Grass-like	Likely yes	Grasslands, forest margins, wet areas	2	1	0	0
Desmodium laxiflorum	Fabaceae		Small shrub	Yes	Forest margins, cultivated	8	1	0	7

Calamagrostis australis	Poaceae (Poeae)	C3*	Grass (perennial)	Yes	Mostly savanna, grassland	0	0	9	0
Dichanthium caricosum	Poaceae (Andropogoneae)	C4*	Grass (perennial)	Yes	Savanna (wetter parts of Asia), tolerant of fire and grazing	0	0	0	7
Engelhardia spicata	Juglandaceae		Tree	Yes	All forest	0	6	0	0
Euphorbiaceae sp	Euphorbiaceae		Small tree	?	?	1	0	0	0
Fabaceae sp	Fabaceae		Small tree	?	?	0	0	1	0
Ficus septica	Moraceae		Tree	Yes	Forest and disturbed forest margins	0	1	0	0
Flacourtia rukam	Salicaceae		Tree	Yes	Forests	3	0	0	0
Gleichenia microphylla	Gleicheniaceae		Fern	Yes	Forests, wet areas	0	8	0	0
Glochidion sumatranum	Phyllanthaceae		Tree	Yes	Forests, forest margins	0	0	1	0
Grewia eriocarpa	Malvaceae		Small tree	Yes	Forests	0	0	1	0
Imperata cylindrica	Poaceae (Andropogoneae)	C4	Grass (perennial)	Yes	Widespread, but often in disturbed areas, fire adapted	0	9	0	0
Lantana camara	Verbenaceae		Shrub	No (central America)	Forests - edges, disturbed	0	2	7	0
Laportea stimulans (syn. *Dendrocnide stimulans*)	Urticaceae		Shrub or Small tree	Yes	Forests, forest margins	0	1	0	0
Melastoma polyanthum	Melastomataceae		Shrub	Yes	Forests, pioneer shrub	0	5	0	0
Morinda citrifolia	Rubiaceae		Tree	Yes	Mostly forests	0	0	0	1
Nauclea orientalis	Rubiaceae		Tree	Likely yes	Forests, pioneer species	0	1	0	0
Ocimum tenuiflorum	Lamiaceae		Forb	Likely yes	Likely forest and cultivated areas	0	4	0	0
Passiflora foetida	Passifloraceae		Climber	No (Americas)	Forests, disturbed	0	0	0	3
Pinanga kuhlii	Arecaceae		Tree (palm)	Yes	Forests (understorey)	0	1	0	0
Polytrias indica	Poaceae (Andropogoneae)	C4*	Grass (Perennial)	Yes	Savanna, grassland, disturbed, cultivated	0	0	0	9
Psidium guajava	Myrtaceae		Tree/shrub	No	Forests, disturbed	1	0	0	0

				(central America)					
Rubus rosifolius	Rosaceae		Climbers and shrubs with thorn	Yes	Forests, disturbed	0	1	0	0
Santalum album	Santalaceae		Tree	No (India)	Cultivated	0	0	1	0
Saurauia javanica	Actinidiaceae		Tree	Yes	Forests, remnants	0	1	0	0
Streblus asper	Moraceae		Tree	Possible yes	Forests, edges	2	0	0	0
Thespesia lampas	Malvaceae		Forb, Small Shrub	Likely yes	Open forests, edges	0	0	0	6
Zoysia matrela	Poaceae (Zoysieae)	C4*	Grass (perennial, stolons)	Likely yes	Coastal, cultivated	0	0	2	0
Ziziphus mauritiana (syn *Z. jujuba*)	Rhamnaceae		Tree (thorny)	Yes	Savanna, open forests, cultivated	0	0	1	2
No. of species						8	14	16	10
Total site occurrences						36	42	51	40

Grass Tribes follow: Soreng, R. J., Peterson, P. M., Romaschenko, K., Davidse, G., Zuloaga, F. O., Judziewicz, E. J., ... & Morrone, O. (2015). A worldwide phylogenetic classification of the Poaceae (Gramineae). Journal of Systematics and Evolution, 53(2), 117-137.

Note: all grass genera listed belong to subfamily Panicoideae (syn. Andropogineae), except *Zoysia* (Chlorodoideae) and *Calamagrostis* (Pooideae)

Grass photosystem from Osborne, C. P., Salomaa, A., Kluyver, T. A., Visser, V., Kellogg, E. A., Morrone, O., ... & Simpson, D. A. (2014). A global database of C4 photosynthesis in grasses. New Phytologist, 204(3), 441-446.

#Native means comes from central Indonesia (Java – Bali – Lombok)
Sources of information :

GBIF	http://www.gbif.org/
ALA	www.ala.org.au

FLORA MALESIANA	http://floramalesiana.org/
PROSEA	http://proseanet.org
BIOPORTAL NATURALIS NL	http://bioportal.naturalis.nl

BOTANICAL GARDEN INDONESIA CATALOGUE

Flora of China

APPENDIX 2: List of species found in chapter 4

List of species found in plots for chapter 4, showing their taxonomic, life-form and habitat characteristics.

Species	Family (Tribes for grasses)	Life-form	Native in study region	Habitat (found in)
Abutilon indicum	Malvaceae	Shrub	Yes	Open forest, dry forests
Acacia leucophloea (syn *Vachellia leucophloea*)	Fabaceae	Tree	Yes	Dry Forest, savanna
Acacia nilotica (syn *Vachellia nilotica)*	Fabaceae	Tree, thorny	No (Africa-Arabia)	Savanna, open forests
Acacia tomentosa	Fabaceae	Tree, deciduous	Yes	Savanna
Acronychia trifoliata	Rutaceae	Tree	Yes	Savanna, dry forest
Ageratum conyzoides	Asteraceae	Herbaceous	No (Central and South America)	Widespread weed open forest, grassland, disturbed areas
Antidesma bunius	Phyllanthaceae	Tree	Yes	Forests
Bauhinia sp	Fabaceae, sub fam Caesalpinioideae	Tree	?	?
Brachiaria ramosa	Poaceae	Grass	No (Tropical Africa) but found also in Asia	Savanna, dry forest, abandoned fields
Capparis sepiaria	Capparaceae	Small tree	Likely yes	Savanna, teak forest
Chromolaena odorata	Asteraceae	Shrub	No (North America)	Widespread weed, forests, edges
Cordia bantamensis	Boraginaceae	Small tree	Likely yes	Dry forests
Cosmos caudatus	Compositae	Herbaceous	No (Latin America) but naturalized SE Asia	Dry forests
Erythrina euodiphylla	Fabaceae	Tree	Yes	Savanna

Flacourtia rukam	Salicaceae	Tree	Yes	Forests
Glochidion sumatranum	Phyllanthaceae	Tree	Yes	Forests, forest margins
Grewia eriocarpa	Malvaceae	Small tree	Yes	Forests
Helicteres isora	Malvaceae	Small tree	Yes	Dry forests
Hibiscus panduriformis	Malvaceae	Shrub	Yes	Savanna
Kleinhovia hospita	Malvaceae	Tree	Yes	Dry forests, savanna, teak forest
Microcos tomentosa	Malvaceae	Small tree	Yes	Moist deciduous and evergreen forests
Mimosa invisa (Syn. *Mimosa diplotricha*)	Fabaceae	Shrub	No (America)	Open forest, dry forest
Oplismenus burmannii	Poaceae	Grass	No (Africa)	Dry forests
Premna tomentosa	Lamiaceae	Tree	Possible yes	Dry forests
Schleichera oleosa	Sapindaceae	Tree	No (Himalayas) but naturalized in Indonesia	Dry forests, savanna
Schlerachne punctata (*Syn. Chionachne punctata*)	Poaceae	Grass	?	Savanna
Schoutenia ovata	Tiliaceae	Small tree	Yes	Dry forests
Streblus asper	Moraceae	Tree	Possible yes	Forests, edges
Tamarindus indicus	Fabaceae	Tree	No (Tropical Africa) naturalized in tropical Asia	Savanna
Ziziphus mauritiana (syn. *Z. jujuba*)	Rhamnaceae	Tree (thorny)	Yes	Savanna, open forests, cultivated

Sources of information :

GBIF	http://www.gbif.org/
FLORA MALESIANA	http://floramalesiana.org/
PROSEA	http://proseanet.org
BIOPORTAL NATURALIS NL	http://bioportal.naturalis.nl
IUCN	http://www.iucnredlist.org/
CABI	http://www.cabi.org/

APPENDIX 3: Link to Published Paper in *Biodiversitas*

Appendix

Fulltext can be freely download from

http://ro.ecu.edu.au/ecuworkspost2013/1537/

http://biodiversitas.mipa.uns.ac.id/D/D1701/D170114.pdf

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Proof of *Acacia nilotica* stand expansion in Bekol Savanna, Baluran National Park, East Java, Indonesia through remote sensing and field observations

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Abstract. *Sutomo, van Etten E, Wahab L. 2016. Proof of* Acacia nilotica *stand expansion in Bekol Savanna, Baluran National Park, East Java, Indonesia through remote sensing and field observations. Biodiversitas 17: 96-101.* One of woody species that is known to inhabit certain savanna ecosystems is *Acacia nilotica*. The *Acacia nilotica* tree is widespread in the northern savannah regions, and its range extends from Mali to Sudan and Egypt. *Acacia nilotica* was first introduced to Java Island in 1850. It then spread to Bali, East Nusa Tenggara, Timor and Papua. Found in grasslands, savanna is reported as important colonizer at Baluran National Park in East Java and Wasur National Park Papua. We conducted Vegetation analysis in three areas of the Baluran Savanna namely: Grazed, burned and unburnt. Our observation result analysis showed that in terms of the three most important tree species in all of the sites that we sampled (grazed, burnt and unburnt savannas) *Acacia nilotica* appeared in each of these sites. The values however, vary between sites. *Acacia nilotica* Importance Value Index is highest in the unburnt savanna, with IVI reaching almost 250. The unburnt site is actually a burnt site but with moderate age or time since fire (approximately 6-7 years), whereas the burnt site is savanna with relatively young age/time since fire (few months to 1 year). We also conducted GIS analysis using Satellite Images (October 2013 and October 2014) to pick up changes in Bekol savanna. Result showed that expansion of *A. nilotica* stand occurred towards the savanna. Over dominance of the woody species *A. nilotica* could shift the savanna into another ecosystem state, i.e. secondary forest.

Keywords: *Acacia nilotica*, Baluran National Park, expansion, remote sensing, vegetation analysis

INTRODUCTION

Before the year 1928, A.H. Loe de Boer, from the Dutch Colonial, had areas of agriculture concession at Labuhan Merak and Mount Mesigit, Baluran. Since then, he had always interested in big mammals and thought that the areas will play important roles in conserving these animals. Following Indonesia independence, the Ministry of Agriculture also stated Baluran as game reserve area with decree No. SK/II/1962. Later on in 1980, the area then stated as a national park, the Baluran National Park with an areas of 25,000 ha. Baluran is known for its vast areas of savanna and is famous as the "Africa van Java" (Baluran National Park 2010).

Savanna is a term to define ecosystem in tropical and subtropical that typical of displaying a continuous herbaceous cover of C_4 grasses that has different patterns based on seasonality in which it is related to water, and where woody plants are also of important values but with sparse patterns and no closed canopy (Frost et al. 1986). Tropical savannas cover over 20% of the Earth surface, with the largest coverage is in Africa, Australia and South America, and just only approximately 10% occur in India Australia (Burrows et al. 1991; Werner 1991), and the Miombo woodland of Southern Africa (Isango 2007). Indonesian savanna however remain somewhat unfamiliar to the scholarly with very few studies have been done. One of the foremost early studies on Indonesian flora was of Backer and van den Brink (1963) and van Steenis (1972). Mountain flora of Indonesia especially in Java Island was what van Steenis specialized. In his report, even in early 1900, grasslands on Java Mountains are already common. Fire, according to van Steenis (1972) was presumed as the major factor that drive the existence of grasslands on Java mountains especially in East Java which is subject to dryer dry season, lower precipitation compare to other parts of the island.

One of woody species that is known to inhabit certain savanna ecosystem is *Acacia nilotica* (Figure 1). *Acacia nilotica* is known to be abundant originally in Africa (Brenan 1983) and has been scantly studied. In a study by Skowno (1999) in Hluhluwe Game Reserve, South African savanna, *A. nilotica* was described in terms of their quantitative structure and distribution. In Australia, this species is dominant in Queensland where it has been declared as weeds, and only a few are found in Western

APPENDIX 4: Link to Published Paper in Tropical Drylands

Fulltext can be freely download from

https://smujo.id/td/article/download/1644/1580/

TROPICAL DRYLANDS
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DOI: 10.13057/tropdrylands/t010106

Species distribution model of invasive alien species *Acacia nilotica* for Central-Eastern Indonesia using Biodiversity Climate Change Virtual Laboratory (BCCVL)

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Abstract. *Sutomo, Van Etten E. 2017. Species distribution model of invasive alien species Acacia nilotica for Central-Eastern Indonesia using Biodiversity Climate Change Virtual Laboratory (BCCVL). Trop Drylands 1: 36-42.* Climate change may facilitate alien species invasion into new areas. This study uses Biodiversity and Climate Change Virtual Laboratory to develop a species distribution model (SDM) of *Acacia nilotica* (L.) Willd. ex Delile. based upon its naturalized distribution to project the potential distribution of *A. nilotica* throughout tropical environment of Indonesia under current and future climate conditions. Global biodiversity information facility database was utilized to obtain the species occurrences data. The climate factors were precipitation and temperature layers, available in Worldclim current conditions (1950-2000) at 2.5 arcmin. We used Generalized Linear Model. The result was then projected to the year 2045 using RCP 8.5 greenhouse gas emissions scenarios to influence the climate model (CSIRO Mark 3.0. with 30'' resolution). Final results show that global climate change is likely to increase markedly the potential distribution of *A. nilotica* in Indonesia. By the year 2045, *A. nilotica* is most likely to spread to eastern parts of Indonesia. In general, our model performance is good (AUC = 0.82), however, like many other SDMs, it does not take into account biotic interactions as well as other environmental factors. Nonetheless, climatic suitability is an essential requirement for successful establishment of an invasive species and species distribution models that can disclose general patterns and convey useful estimate.

Keywords: *Acacia nilotica*, invasive alien species, species distribution model, climate change

Abbreviations: GCM = Global Climate Change Model, SDM = Species Distribution Model, BCCVL = Bidoversity and Climate Change Virtual Laboratory, GBIF = Global Biodiversity Information Facility, GLM = Generalized Linear Model, AUC = Area Under the Curve, ROC = Receiver-Operating Characteristics, TPR = True Positive Rate, TNR = True Negative Rate, HSI = Habitat Suitability Index, IAS = Invasive Alien Species, NTT = Nusa Tenggara Timur

INTRODUCTION

Acacia nilotica (L.) Willd. ex Delile is widespread in the northern savanna regions of Africa and its range into the surrounding forests. However, Djufri (2004) found that the existence of Baluran Savanna is threatened by the rapid spread of *A. nilotica*, which has caused changes in some areas of the national park from open savannas to the

APPENDIX 5: Co-Author (of published papers) Statements

To Whom It May Concern,

I, SUTOMO, contributed:

- 80% to the published paper entitle :
  Sutomo, van Etten E. & Wahab L. (2016) Proof of *Acacia nilotica* stand expansion in Bekol Savanna, Baluran National Park, East Java, Indonesia through remote sensing and field observations. *BIODIVERSITAS* **17**, 96-101.

- 80% to the published conference proceeding paper entitle:
  Sutomo, van Etten E. & Priyadi A. (2015) Do Water Buffalo Facilitate Dispersal of Invasive Alien Tree Species *Acacia nilotica* in Bekol Savanna Baluran National Park? In: *Second International Conference on Tropical Biology* (eds E. K. Damayanti and J. C. Fernandez) p. 155. SEAMEO BIOTROP, Bogor.

- 80% to the published paper entitle:
  Sutomo & van Etten E. (2016) Effect of fire and digestion by herbivores on seeds of the exotic invasive species *Acacia nilotica* from savanna at Baluran national parks Indonesia. *Australasian Plant Conservation* **25**, 20-1.

- 80% to the accepted paper entitle:
  Sutomo & van Etten. Species Distribution Model of Invasive Alien Species *Acacia nilotica* (L.) Willd. ex Delile for Central-Eastern Indonesia using Biodiversity Climate Change Virtual Laboratory (BCCVL). *Tropical Dryland.*

I, as a Co-Author, endorse that this level of contribution by the Candidate indicated above is appropriate.

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